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Back cover photos: Arctic Tern: Callum Macgregor; Rook: Annie Haycock; Red Kite: Tony Pope

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Editorial

Dear readers

It is not an easy task to take on something as complex as a scientific journal. Every edition is written by multiple authors, some experienced and scientifically trained, others bringing a lifetime of passion or decades of field observation. The Editor's primary task is to welcome and shape all those different backgrounds, voices and stories into articles for publication. I did not - could not - do this job alone, and would like to thank our authors, the very supportive WOS Council, the Editorial Board and in particular our enormously experienced type-setter for catching my mistakes, teaching and encouraging me through the last eight months.

In this issue we begin with an unusual but important supplement to the publication of Birds of Conservation Concern Wales 4 (Johnstone *et al.* this journal 2023), presenting assessments for Wales' 21 breeding seabirds. The report uses the complete datasets from the 'Seabirds Count' surveys (2015-21) and additional data collected during the major outbreaks of highly pathogenic avian influenza (HPAI) in 2022 and 2023. Our cover image highlights the major impact of HPAI on Gannet, which becomes the eighth seabird species on the Welsh Red list.

I am also pleased to bring together two papers presenting and celebrating both the delivery and results of the 2022-23 Rook Survey. Successful national-scale volunteer surveys require not only large numbers of people, but also a wide range of skill sets from the field to the analytical backroom. The author and I hope that the first paper will both inspire and assist the coordinators of the next all-Wales species survey in their task. The second paper presents the complete analytical outputs of the survey, highlighting complex changes in land use, climate, tree species and Rook colony distribution that are impacting Rook populations across Wales.

You will have noticed the different print format of this year's Milvus. We have returned to the smaller A5 print size, primarily because printing *Milvus* in this format very considerably reduces its environmental footprint. Perhaps a less obvious change - but one I hope to see grow and develop through future editions - is the inclusion of more short articles. I look forward to including interesting notes, intriguing negative (or non-significant?) results; perhaps observations from local studies and certainly more citizen scientists. *Milvus* should be a journal that brings Wales' scientists, professionals and citizen scientists together through our shared interest in Welsh birds. I and the Editorial Board are here to help and advise you through every step of the process - bring us your best ideas!

Rachel Taylor

"Adar o'r unlli, ehedant i'r unlle"

Rooks are easily recognised, noisy and social corvids, differentiated from the similarly sized Carrion Crow by their harsh 'rook!' call, somewhat shaggy 'trouser' plumage and their distinct pale grey face (created by bare facial skin and the grey base to the tapered bill). Breeding Rook are characteristically colonial, building large, closely spaced, twiggy nests in tall trees - often on the edges of woodland, in shelterbelts or mature hedgerows. Most nesting trees are broadleaved species, historically dominated by English elm but more recently a wider range of broadleaves and occasionally conifers (Brenchley, this volume). These 'rookeries' can contain as many as 500 nests, and are occupied from as early as February in a mild year, but more usually March to June. On average each pair lays three or four eggs

and fledges two chicks (BTO 2024). Rooks are faithful both to their partner and to the nest itself and remain social year-round. In winter they may also share large roosts with other corvid species.

Rooks' systematic name, *Corvus frugilegus*, translates as 'fruit-gatherer', and diet studies show that over half their diet is plant material (Waite, 1985). There is some seasonal and geographic variation in Rook diet, with soil invertebrates particularly important in the breeding season. A wide range of hedgerow fruits and berries, seeds, root crops and (less commonly) cereals contribute almost 60% of their annual average food intake. Rooks are significant seed dispersers in some landscapes (Kitowski *et al.* 2017), and have been recorded feeding opportunistically in a broad range of places including on landfill sites, intertidal habitats and even roadkill. Their strong association with agricultural landscapes has repeatedly led to their being perceived as agricultural pests in Britain.

Rook have been both intensely surveyed and commonly subjected to control and persecution across the UK, likely as long as agricultural land use has altered British landscapes. Certainly their survey history is almost as long as that of the BTO itself; starting with the Oxford district survey instigated by E.M. Nicholson and delivered by local birdwatchers in 1927-28. More surveys followed: the upper Thames in 1931, Gloucestershire and Somerset in 1933 and then a Ministry-commissioned national survey in 1944-46. Contrary to Government concerns at the time, Rooks were more associated with grassland than crops, although 80% of the food found in sampled birds was vegetable matter. For the next two or three decades county bird reports indicated that Rook populations continued to increase, but in the 1960s some areas began to report marked declines and a second national survey was launched in 1975-77. Rooks had declined by 30-60%, driven by urbanisation, loss of grassland and nesting habitats (Sage and Vernon 1978). A repeat sample of this survey in 1980 found further changes: fewer but slightly larger rookeries, changes in tree species choice and further pressure from agriculture (Sage and Whittington 1985). By 1996, after three years of licensed control, another national survey intended to revise the national population estimate and trend found that Rooks had increased 41% since 1975 but not recovered to their 1940s high point (Marchant and Gregory 1999).

The Welsh Rook Survey adds significantly to this rich history of population changes driven by complex interactions between human activity, this social and adaptable species and the British landscape. Rooks have been impacted both positively and negatively by agricultural changes, but the negative pressures are perhaps becoming more widespread. They have been challenged by two novel tree diseases (Dutch Elm Disease and Ash Dieback), culled and persecuted, subjected to the complex pressures of agricultural intensification and, increasingly, climate change. Perhaps it is time to celebrate the persistence of our fruit-gathering crow, and use our updated understanding of its Welsh distribution to support its resilience and ours.

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The status of breeding seabirds in Wales: an addendum to the fourth assessment of Birds of Conservation Concern Wales

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Abstract
This paper supplements the fourth assessment of the population status of birds in Wales, published in 2022. Assessing the status of birds between the six-year cycles of Birds of Conservation Concern is unprecedented but necessitated by the availability of final data from a major census of breeding seabirds in Britain and Ireland (2015-21) and additional surveys undertaken in 2023, in response to the apparent significant impact of Highly Pathogenic Avian Influenza (HPAI) on seabirds.

Data on breeding seabird populations were assessed against standardised criteria, in a two-stage process using data from monitoring and surveillance schemes in different time-series. Each species was assigned to Green, Amber, or Red lists, indicating increasing levels of conservation concern. A total of 21 species was assessed. Gannet *Morus bassanus* was elevated to the Red list for the first time due to a sudden population decline caused by HPAI. The concern rating for Black Guillemot *Cepphus grylle* was lowered to Green and that for Lesser Black-backed Gull *Larus fuscus* was corrected to Amber and remains unchanged since BoCCW3 in 2016. Our results show that eight of the 21 breeding seabirds are Red-listed and 11 are Amber-listed, and so seabirds remain the group with the highest levels of conservation concern in Wales. The country's seas and coast continue to be of international importance for four species. Of all Welsh birds, there are 60 species (27%) on the Red list, 90 (41%) on the Amber list and 70 (32%) on the Green list.

Crynodeb
Mae'r papur hwn yn ategu'r pedwerydd asesiad o statws poblogaeth adar yng Nghymru, a gyhoeddwyd yn 2022. Mae asesu statws adar rhwng cylchoedd chwe blynedd Adar o Bryder Cadwraethol yn ddigynsail, ond mae'n ofynnol oherwydd bod data ar gael o gyfrifiad mawr o adar môr sy'n bridio ym Mhrydain ac Iwerddon (2015-21) ac arolygon ychwanegol a gynhaliwyd yn 2023, mewn ymateb i effaith sylweddol ymddangosiadol Ffliw Adar Pathogenig lawn (HPAI) ar adar y môr.

Cafodd data ar boblogaethau adar môr sy'n bridio ei asesu yn erbyn meini prawf safonol, mewn proses dau gam gan ddefnyddio data o gynlluniau monitro a gwyliadwriaeth mewn cyfresi amser gwahanol. Cafodd pob rhywogaeth ei neilltuo i restrau Gwyrdd, Oren neu Goch, sy'n dangos lefelau cynyddol o bryder cadwraethol. Gyda'i gilydd aseswyd 21 rhywogaeth. Aeth y fulfran wen *Morus bassanus* i fyny i'r Rhestr goch am y tro cynaf oherwydd gostyngiad difrifol mewn poblogaeth oherwydd HPAI. Cafodd sgôr pryder y Wylog Ddu *Cepphus grylle* ei ostwng i Gwyrdd a chafodd sgôr y Wylan gefnddu leiaf *Larus*

fuscus, ei gywiro i Oren, ac mae'n aros yr un fath ers Adar o Bryder Cadwraethol 3 yn 2016. Mae ein canlyniadau'n dangos bod wyth o'r 21 o adar môr sy'n bridio ar y rhestr Goch a bod 11 ar y rhestr Oren, felly adar môr yw'r grŵp sydd â'r lefelau uchaf o bryder cadwraethol yng Nghymru o hyd. Mae moroedd ac arfordir y wlad yn parhau i fod o bwysigrwydd rhyngwladol i bedair rhywogaeth. O holl adar Cymru, mae 60 o rywogaethau (27%) ar y rhestr Goch, 90 (41%) ar y rhestr Oren a 70 (32%) ar y rhestr Werdd.

Introduction
The fourth assessment of 'Birds of Conservation Concern in Wales' (BoCCW4) was published in 2022 (Johnstone *et al.* 2022). The authors recognised that the data on abundance of breeding seabirds in Wales were provisional and recommended that the status of those species be reviewed once the full results of the Seabirds Count 2015-21 census of Britain and Ireland were published. The findings from Seabirds Count are now published (Burnell *et al.* 2023) and present changes in 28 UK breeding seabird species since Seabird 2000 (Mitchell *et al.* 2004). In the interests of consistency and stability, the multi-partner group of statutory and non-government organisations that produce BoCCW generally avoid publishing updates between the six-year cycles. However, the impact of the H5N1 strain of Highly Pathogenic Avian Influenza (HPAI) on seabirds around northwest Europe during 2022 prompted further surveys in 2023 of those species at greatest risk. The availability of these datasets and subsequent addendum of the fifth assessment of 'Birds of Conservation Concern in the UK' (BoCCUK5, Stanbury *et al.* 2024) resulted in the decision to publish this revised assessment for the breeding status of seabirds in Wales (hereafter termed BoCCW4a) ahead of BoCCW5, planned for 2028. Further background is provided in Stanbury *et al.* (2024) and Johnstone *et al.* (2022).

Methods
The BoCCW4 review (Johnstone *et al.* 2022) assessed bird populations against standardised criteria relating to population status. This addendum (BoCCW4a) encompasses the 21 seabird species that breed in Wales and assessed their status against the five relevant criteria (11 sub-criteria). In line with BoCCW4, it did not assess races separately and did not assess non-breeding populations, since no new data were available.

Meeting one or more criterion qualifies a species for the relevant list, with species being placed on the highest priority list for which they qualify (i.e. those qualifying against a Red-listed criterion will be placed on the Red list regardless of qualification against an Amber-listed criterion). Species that met none of the criteria because of, for example, population increase or stability, along with any that have continued to recover from historic decline will be placed on the Green list.

The criteria
The BoCCW4 criteria reflect a hierarchical approach to geographical scale, determining population status based on: global importance, historical population decline, recent population decline, European importance, breeding rarity, localised distribution, UK importance and international importance. The 11 sub-criteria used BoCCW4a (Table 1) are those where there was new information since BoCCW4; greater detail about the approach was published in Johnstone *et al.* (2022).

Species could still qualify for listing on one of the 19 other criteria that did not relate to breeding numbers, and these are unchanged since the 2022 assessment.

Criterion 2. **HD**, **HD_{REC}** **HD_{REC2}**: HISTORICAL DECLINE IN BREEDING POPULATION.

Criterion 3. **BD**, **BDM**: RECENT AND LONGER-TERM BREEDING POPULATION DECLINE. For each sub-criteria (time-period and abundance/range), we distinguish between three levels of change: rapid (at least 50% decline) and moderate (at least 25% but less than 50% decline) and stability (less than 25%

decline) to

List	Criteria ¹	Name	Definition	Abbrev	First use ²
Red	3	Recent population decline	≥50% decline in breeding population (20 years)	BDp1	1
Red	3	Recent population decline	≥50% decline in breeding population (50 years)	BDp2	2
Amber	2	Historic decline – recovery	Was Red but population increase of greater than 100% in last 20 years	HD rec	1
Amber	3	Recent population decline	≥25% but ≤50% decline in breeding population (20 years)	BDMp1	1
Amber	3	Recent population decline	≥25% but ≤50% decline in breeding population (50 years)	BDMp2	2
Amber	5	Breeding rarity	Mean of <30 breeding pairs during 2015-21	BR	1
Amber	6	Localised breeding	≥50% of population occurs at one site, but not a rare breeder	BL	1
Amber	7	UK	Red-listed at a UK-level	UKRed	1 ³
Amber	7	UK	≥50% of UK breeding population occurs in Wales	UK ₅₀	2 ³
Amber	7	UK	≥10% of the population of a UK rare breeding species occurs in Wales	BR _{UK}	2 ³
Green	2	Historic decline – recovered	Previously Amber and at least 167% increase since first HDrec qualification	HDrec2	1

Table 1. Assessment criteria used in the seabird addendum for BoCCW4.

¹ Criteria not listed here are in Table 1 of Johnstone et al. 2022. ² BoCCW version of first use. ³ Used for Wales

distinguish between qualification for Red, Amber or Green lists. Data from the last 20 years (Mitchell *et al.* 2004) and 50 year (Cramp *et al.* 1974) were used to reflect species whose decline lies between historical and recent, and from which there has been no recovery.

Criterion 5. **BR**: BREEDING RARITY. Species were categorised as rare breeders in Wales if they had a breeding population of fewer than 30 pairs.

Criterion 6. **BL**: LOCALISED POPULATIONS. Populations that are geographically concentrated are assumed to face greater threats from chance events than those that are more dispersed. Rare breeders were not assessed against this criterion as their small numbers and range make them more likely to be localised. Amber listing under the localised criterion is intended to signal a species’ vulnerability, as local pressures (e.g., disease or an oil pollution incident) could adversely impact a large proportion of the population.

The criterion was based on the single best site, either a Special Protection Area (SPA) and/or Important Bird Area (IBA). Species with 50% or more of their population at a site qualified for the Amber list. Data for the most populous site in the breeding and non-breeding seasons were compared with Wales’s population estimates for the same period.

Criterion 7. **UKRed/UK₅₀/BR_{UK}**: UK IMPORTANCE. Red-list qualification at the UK-level (Stanbury *et al.* 2024) is used as an Amber-list qualification in Wales. This ensures that UK priorities are fully considered at the Wales-level in the same way that European priorities are considered in the UK and Welsh reviews. Wales may support substantial proportions of the populations of some UK species with a western

distribution, and consequently sub-criteria show when Wales has 50% or more of the UK population of a species, and 10% or more of a UK rare breeding species. The values of these thresholds were pragmatic, but consistently applied across successive reviews.

Data sources

The assessment took a two-stage approach using data: (i) for all breeding seabirds up to 2015-21 (pre-HPAI); (ii) from 2023 for species where there was reliable data. Scientific names for species are provided in Table 2.

Stage 1. Final Seabirds Count population estimates (pre-HPAI)

The principal sources of data for stage one were the four seabird censuses 1969–70 (Cramp *et al.* 1974), 1985–88 (Lloyd *et al.* 1991), 1998–2002 (Mitchell *et al.* 2004) and 2015–21 (Burnell *et al.* 2023). Provisional data from Seabirds Count 2015-21 were used in the 2022 assessment of BoCCW4. Finalised data are now available for all breeding seabird species in both Wales and UK (Burnell *et al.* 2023), with population estimates showing changes for some, both increase and decrease. This is due to counts being screened for duplicate data and, for terns that are counted annually, Burnell *et al.* (2023) selecting a different year’s population estimate to that used in BoCCw4. Importantly, UK population estimates, previously missing for many species, are now available.

Not all assessment criteria needed to be reviewed to incorporate revised Seabirds Count data. Those that do include breeding population decline over the two time periods (**BDp1** and **BDp2**) in the event that 25 year or longer term percentage declines now meet different threshold values. No longer qualifying under breeding rarity (**BR**) is also possible.

Moreover, some BoCCW4 criteria use information from the fifth UK BoCC assessment (Stanbury *et al.* 2024), and therefore it is possible that listings in Wales have been influenced by changes to the UK list and its underlying data. For example, any UK Red-listed species that occurs in Wales qualifies as Amber here (**UKRed**). Also, the Welsh population size as a proportion of the UK population is used to identify species in Wales with UK importance (Amber qualifying under **UK₅₀**), and so revised UK abundance data could influence whether species meet this criteria in Wales.

Importantly, although revised Seabirds Count data may result in changes in qualification under one or more criteria, this will only affect overall qualification of a species if it results in a higher or lower qualification across all criteria. For example, a seabird that qualified Amber in BoCCW4 but now qualifies Amber on additional criteria in BoCCW4a will not change its overall level of conservation concern.

Stage 2. Apparent HPAI impacts to date

The principal additional data source for stage two was Tremlett *et al.* (2024). Surveys were undertaken of 10 of the 21 breeding seabird species in 2023 as part of a UK-wide assessment of the impact of HPAI during the previous breeding and non-breeding seasons. Our Stage-2 assessment did not take into account UK data and used abundance data only where coverage was >70% of the Seabird 2000 census in Wales, i.e. for Black-headed Gull, Sandwich Tern, Little Tern, Arctic Tern, Common Tern, Roseate Tern and Gannet.

For those seven species for which there was adequately representative data in Stage 2, the 2023 figure was used. For the remainder, the 2015-21 figure (from Stage 1) was used.

Results

Stage-1 assessment: change based on final Seabirds Count estimates (pre-HPAI)

In total, 21 Welsh breeding seabird species were assessed for inclusion in BoCCW4a. Qualification on

Table 2. Results of the BoCCW4a revision using finalised Seabirds Count data in comparison with previous assessments. Overall changes between BoCCW4 and BoCCW4a are in black squares. Gannet was the only species affected by the Stage-2 assessment. Note that the ‘change’ for Lesser Black-backed Gull results from a data error – which incorrectly placed it on the Red list in BoCCW4 – as discussed in the text.

Common and HBW/BirdLife International scientific names	Previous BoCCW reviews					BoCCW4a qualifying criteria ¹
	BoCCW1	BoCCW2	BoCCW3	BoCCW4	BoCCW4a	
Arctic Tern <i>Sterna paradisaea</i>	A	R	R	R	R	BDr2 BL UKRed HDrEc2 UKRed
Black Guillemot <i>Cepphus grylle</i>	A	A	A	A	G	(BDMp1) (BR)
Black-headed Gull <i>Chroicocephalus ridibundus</i>	A	R	R	R	R	BDp2 BDMr2 WI
Common Tern <i>Sterna hirundo</i>	A	R	A	A	A	BDMr2 BL
Cormorant <i>Phalacrocorax carbo</i>	A	A	A	G	G	
European Storm-petrel <i>Hydrobates pelagicus</i>	A	A	A	A	A	BL
Gannet <i>Morus bassanus</i>	A	A	A	A	R	BDp1 BL BI
Great Black-backed Gull <i>Larus marinus</i>	A	R	R	A	A	BDMp2 UKRed
Guillemot <i>Uria aalge</i>	G	A	A	A	A	BI
Herring Gull <i>Larus argentatus</i>	A	R	R	R	R	BDp2 BDMp1 UKRed WI
Kittiwake <i>Rissa tridactyla</i>	A	G	R	R	R	IUCN ERLob BDMp1 BDMp2 UKRed
Lesser Black-backed Gull <i>Larus fuscus</i>	A	A	A	A	A	BDMp1
Little Tern <i>Sternula albifrons</i>	R	R	R	R	R	BDr1 BDr2 BL HDrEc2
Manx Shearwater <i>Puffinus puffinus</i>	A	A	A	A	A	BDMr2 UK50 BI
Mediterranean Gull <i>Ichthyophaga melanocephalus</i>	G	A	G	A	A	BR
Northern Fulmar <i>Fulmarus glacialis</i>	G	G	A	A	A	ERLOB BDMp1
Puffin <i>Fratercula arctica</i>	R	R	R	R	R	IUCN ERLob UKRed HDrEc2
Razorbill <i>Alca torda</i>	G	G	A	A	A	BI
Roseate Tern <i>Sterna dougalli</i>	R	R	R	R	R	BDp1 BDp2 BDMr1 BR UKRed
Sandwich Tern <i>Thalasseus sandwicensis</i>	A	A	A	A	A	BDMr2 BL
Shag <i>Phalacrocorax aristotelis</i>	G	G	A	A	A	BDMp1 (UKRed)

1. Additional qualifying criteria resulting from BoCCW4a in bold; removal of qualifying criteria in parentheses

individual criteria was affected in five species (Great Black-backed Gull, Lesser Black-backed Gull, Arctic Tern, Black Guillemot and Shag) under three different criteria (see Table 3). Changes in the UK Red-list influenced three species in Wales (Great Black-backed Gull, Arctic Tern and Shag).

The change for Lesser Black-backed Gull resulted from an error in the data used from Seabird 2000, which caused it to be attributed, incorrectly, to the Red list in 2022. Thus, the only species for which the overall level of conservation concern was affected (owing to an increase in breeding numbers) was Black Guillemot, which moves from Amber to Green (Table 2). These changes result in only minor changes in the number of species on the revised list, reducing the Amber List by one and increasing the Green List by one (Fig.1). A consolidated list is published on the WOS website <https://birds.wales/wp-content/uploads/2024/09/BoCCW4-consolidated-list-September-2024.pdf>

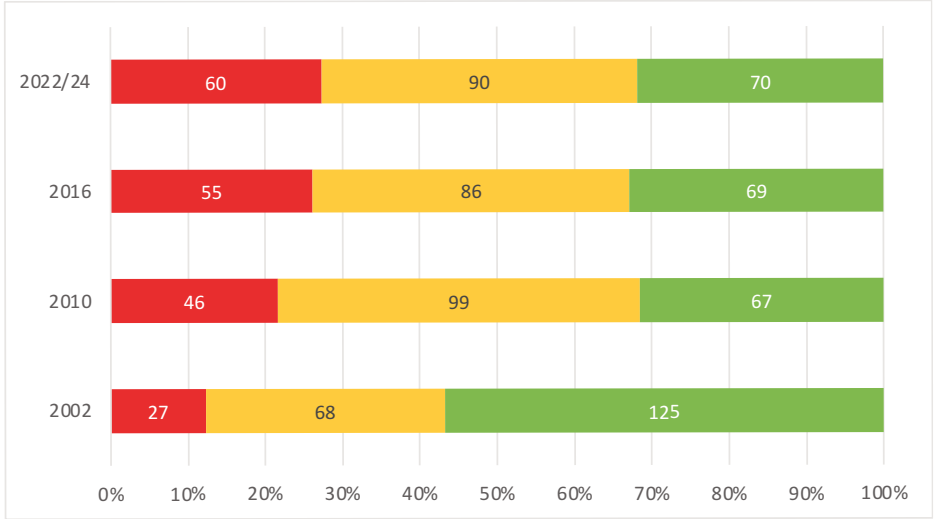


Figure 1. The proportions of assessed species qualifying for each list in successive iterations of BoCCW4 (BoCCW4 following the seabird addendum) annotated with the number of species on each list.

Stage-2 assessment: Impact of HPAI

The outbreak of HPAI in UK seabirds began in 2021 and spread to Wales by August 2022. Of the seven species assessed at stage 2, only Gannet changed its level of conservation concern. Black-headed Gull, Common Tern and Sandwich Tern showed evidence of Red- or Amber-qualifying magnitudes of decline, but their overall status was unaffected (Table 3).

Discussion

The increased level of concern for Gannet resulted from a reduction from 30,688 Apparently Occupied Sites on Grassholm in 1999 to 16,482 in 2023, the smallest population since 1969. The 1999-2023 decrease was 46%, below the threshold for Red-listing. However, the 54% decline between 2022 and 2023 was considered sufficient to be placed on the Red list. Prior to HPAI, Grassholm held c.7% of the global Gannet population and was the fourth largest colony in the world. A second Gannet colony on Anglesey, which is still in the early stages of establishment, also suffered mortality from HPAI.

The HPAI virus appears to have affected populations of other species, particularly Black-headed Gull,

Sandwich Tern and Common Tern, at least in the short-term. It remains unclear how seabird population growth will respond to this event in subsequent years. Robust levels of annual monitoring of these species in Wales will be necessary to assess changes with a high degree of confidence in the fifth assessment of BoCC Wales, planned for 2028.

There was a reduction in the level of concern for Black Guillemot (Amber to Green) and Lesser Black-backed Gull remains (from BoCCW3 in 2016) as Amber-listed, correcting for the error in Johnstone *et al.* 2022. The concern rating for Lesser Black-backed Gull would have been Green had the number of roof-nesting pairs been included in the total, but since there was no comparable census of the roof-nesting population in Wales prior to Seabirds Count (2015-21), we have followed the convention adopted by Stanbury *et al.* (2024) at a UK level and only used comparable data from natural nest sites.

Changes in abundance in Wales may have affected qualification for international importance (BI), but there is no recent comprehensive Europe-wide population estimate post HPAI.

While numbers of most seabird species in Wales showed less severe decline than in other parts of the UK, particularly Scotland, the assessment serves as a reminder of the vulnerability of some populations. Of the 21 Welsh breeding species, eight declined between Seabird 2000 and Seabirds Count (2015-21): Fulmar, Cormorant, Shag, Kittiwake, Black-headed Gull, Lesser Black-backed Gull, Herring Gull and Roseate Tern. Seven species breed at three sites or fewer: Gannet, Mediterranean Gull, Little Tern, Sandwich Tern, Roseate Tern, Arctic Tern and Black Guillemot. Of those whose populations increased, many have done so owing to dedicated wardening effort at colonies (e.g. Sandwich Tern, Little Tern, Arctic Tern) and biosecurity of the principal island colonies that has maintained their rodent-free status (e.g. Puffin, Manx Shearwater).

The UK remains internationally important for its breeding seabird assemblage, and Wales has a particular global responsibility. Our offshore islands host up to 64% of the World's Manx Shearwater and 7% of Gannet in the breeding season. Wales also holds breeding populations of Razorbill and Guillemot that are important in an international context. This assessment, like the initial BoCCW4, did not separately consider the geographic races of each species, principally because not all can be differentiated in the field. However, there are some cases where Wales is important at race level. For example, almost all of the Welsh breeding population of Cormorant is of the Atlantic race, *P.c. carbo*, of which Wales holds more than 2% of the population.

It is anticipated that climatic change will provide seabirds, and their prey, with major challenges in the coming decades. Making populations, of both breeding and wintering seabirds, more resilient to this should be a key feature of a Wales Seabird Conservation Strategy to which Welsh Government has committed.

Acknowledgments

We thank the wardens of nature reserves and bird observatories that hold the majority of Wales' breeding seabirds, and volunteers at other sites who contribute data to the BTO/JNCC Seabird Monitoring Programme (bto.org/smp).

Ethical statement

This report is based on data collected by experienced bird watchers and ornithologists who were careful not to cause disturbance to the birds they were monitoring.

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Gannet: Callum Macgregor

The WOS Breeding Rook (*Corvus frugilegus*) Survey 2022-23: a case study for single-species volunteer surveys

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Abstract

The 2022-23 Wales Rook survey was a major single-species sample survey with national coverage. It was initiated, organised and delivered by WOS over two field seasons with advice and support by conservation partners (NRW, BTO and RSPB). Simultaneous full (census) coverage was undertaken in some counties.

In the process of building, delivering and analysing the data from this survey, the organisers developed an improved understanding of what is needed to deliver such a survey in terms of the critical steps and decisions involved. We present a planning and design structure for future Welsh surveys, including a suggested framework for choosing a species suited to volunteer surveys. The design phase of the 2022-23 Rook survey is presented as a case study using this structure.

The field survey covered 1027 tetrads using at least 350 volunteers, generating 2356 unique online records. Rookeries averaged 23 nests, having declined from 25 nests in 1975. Most rookeries were in broadleaved trees; notably oak, ash, beech and sycamore. We present summaries of rookery size data, tree species data and geographic distribution and discuss the likely pressures on Rook populations in contemporary landscapes.

Finally, we discuss a series of lessons learned from this survey with clear application to the design, planning and delivery of single-species all-Wales surveys organised by WOS in future.

Crynodeb

Roedd Arolwg Ydfran Cymru 2022-23 yn arolwg sampl sylweddol o un rhywogaeth ar raddfa genedlaethol. Cychwynnwyd, trefnwyd a gweithredwyd yr arolwg gan CAC dros ddau dymor, gyda chyngor a chefnogaeth gan bartneriaid yn y maes cadwraeth (CNC, BTO a'r RSPB). Ar yr un pryd cynhaliwyd arolwg llawn (cyfrifiad) mewn rhai siroedd.

Yn ystod y broses o adeiladu, gweithredu a dadansoddi'r data o'r arolwg, datblygodd y trefnwyr well dealltwriaeth o'r hyn oedd yn angenrheidiol ar gyfer y fath arolwg o ran y camau hanfodol a'r penderfyniadau oedd yn rhan o'r gwaith. Rydym yn cyflwyno strwythur cynllunio a dylunio ar gyfer arolygon yng Nghymru yn y dyfodol, yn cynnwys awgrym am fframwaith ar gyfer dewis rhywogaeth addas ar gyfer arolwg gan wirfoddolwyr. Cyflwynir y gwaith cynllunio ar gyfer Arolwg Ydfran 2022-23 fel esiampl yn defnyddio'r strwythur yma.

Archwiliodd yr arolwg maes 1027 o detradau, gan ddefnyddio o leiaf 350 o wirfoddolwyr a chynhyrchu 2356 o gofnodion ar-lein unigryw. Ar gyfartaledd roedd 23 nyth ymhob nythfa, gostyngiad o'r 25 nyth yn 1975. Roedd y mwyafrif o'r nythfeydd mewn coed llydanddail, yn enwedig derw, onnen, ffawydden a sycamorwydden. Rydym yn rhoi crynodebau o ddata ar faint nythfeydd, rhywogaeth y coed a dosbarthiad daearyddol, ac yn trafod y pwysau tebygol ar boblogaethau Ydfran mewn tirluniau modern. Rydym yn gorffen trwy drafod cyfres o wersi a ddysgwyd o'r arolwg y gellir eu defnyddio wrth ddylunio, cynllunio a gweithredu unrhyw arolygon o un rhywogaeth a drefnir gan CAC yn y dyfodol.

Introduction to surveys

Every survey is designed for a specific purpose, and bird surveys can generally be assigned to one of two broad categories; those that measure change and those that estimate population size. Each requires very different approaches in methodology, coverage and timescale. A survey designed to monitor change may not be capable of delivering a population estimate (or vice versa). Change measures require consistent survey methods, adequate spatial coverage and at least one repeat, while population estimates require greater coverage of the target population and consideration of targeting and analytical approaches capable of compensating for incomplete coverage, species detectability and differences between surveyors, in addition to any important ecological variables (Bibby *et al.* 2000).

National monitoring schemes – surveys for measuring change

Common breeding birds are monitored by the BTO/JNCC/RSPB Breeding Bird Survey (BBS), a UK-wide, annually repeated partnership survey jointly funded by the BTO, RSPB and JNCC, with fieldwork conducted by volunteers. This survey underpins the UK Government's Wild Bird Indicator. In Wales, the BBS covers more than 300 1-km squares and analysis delivers annual population trends for the 60 species occurring in at least 30 BBS squares each year. Change metrics derived from BBS are generally robust for the species reported, but may not reflect changes in species ecology (for example colony dynamics).

For species of more restricted range, very specific habitat requirements, small populations or challenging ecology, different approaches are required. Seabirds, for example, are monitored at colony level, using a specific suite of methods.

Surveys to estimate (or census) population

UK wide single species surveys may be required for statutory purposes and/or where a species is of high or increasing conservation concern, and/or is not well served by other monitoring scheme methods (including spatial or habitat coverage, or key ecological variables such as colony size). These have been supported by Statutory Nature Conservation Bodies (in Wales this is Natural Resources Wales (NRW)) and JNCC. Most such surveys are organised by the BTO or RSPB and many delivered via BTO's regional network of volunteers. For challenging terrestrial species including some raptors, the SCARABBS (Scarce and Rare Breeding Bird Surveys) programme of UK surveys of such species is managed by the Statutory Nature Conservation Bodies and the RSPB, and undertaken by volunteers often supplemented by paid surveyors. These meet the design and coverage requirements to produce population estimates, but when repeated can also measure change.

Most recently, two UK surveys (House Martin *Delichon urbicum* in 2016/17 and Tawny Owl *Strix aluco* in 2018/19) included significant coverage in Wales. However, to date the only all-Wales survey organized by the BTO was the Welsh chat survey of 2012 and 2013 which included three species (Wheatear *Oenanthe oenanthe*, Whinchat *Saxicola rubetra* and Stonechat *Saxicola rubicola*) allied by similar habitat requirements (Henderson *et al.* 2017).

Single-species surveys may also be planned and delivered at sub-national spatial scales, coordinated and delivered by local bird clubs or individuals.

In the last decade, fewer single species surveys have been delivered, likely as a result of decreased funding from governmental bodies and the impact of the Covid -19 pandemic. In the face of these challenges, conservation and monitoring effort has been focused towards maintaining effort on core monitoring surveys, such as BBS and the Wetland Bird Survey (WeBS).

Delivering a single species survey

There are two key parts to undertaking a single species survey; the choice of species and the design of

the survey itself, to ensure it is capable of delivering its intended outcomes.

Species choice will depend on a range of factors including scientific aims, real-world practicality and degree of urgency. This can be framed as a series of questions, such as:

Why does a species need specific survey?

Conservation concern is a prime reason; for example a species' status degrading from green to amber, amber to red or even green to red. This could be due to a species' range retracting or contracting and whether this is a key species in a particular region (e.g. importance in Wales). Another possibility is poor coverage under established monitoring schemes; or existing schemes (such as BBS), not being tailored to the ecology of a particular species.

Who wants to know?

Whilst adding to our knowledge about any bird species is always desirable, undertaking widespread and detailed survey is usually the result of its importance to key decision makers - in this case primarily the Statutory Nature Conservation Body (in Wales, NRW), along with key partners BTO and RSPB. Here, survey design considerations need to take into account whether this is a baseline survey (for future change analysis) or a stand-alone population estimate? Does it need to be robust enough to underpin significant conservation decision making?

Survey by volunteers or professionals – things to consider

Can volunteers get to where the species is, how much walking is required for the survey method, are there access restrictions to consider? What skill level/standard of optics or other equipment is required for confident identification? When is the species present, is it cryptic, does its detectability change quickly (or predictably) in-season? What level of permission (e.g. Schedule 1 licence) is required for the appropriate survey method? How rewarding is the survey for participants?

Engagement and up-skilling has benefits too

Surveys provide the opportunity for new volunteers to take part and whose newly discovered skills might then be directed into other surveys, including the national monitoring schemes. Existing volunteers may be enthused to continue their interest in a particular species and do more local long-term monitoring or engage with practical conservation activities.

Support and funding

A common misconception around citizen science monitoring schemes is that they can be undertaken for free, because the surveyors are volunteers. However, the long-term monitoring schemes listed above, although delivered by volunteers, are designed, supported, analysed and reported by professional scientists with experience of statistical modelling. For most surveys, some key tasks (survey design, coordination, analysis and reporting, development and support of online tools) require very significant and time-consuming input from people with considerable skills in order for the survey to meet its aims; and this time and skill may not be readily available from volunteers. It is important to recognise these costs, and consider how much support the survey will need and where funding might be found if financial support is required.

Having decided on a species, we move into the survey planning phase, which has several key stages. Critically, these include enlisting support from key organisations for scientific advice and funding, identifying means of coordination, engaging with a suitable workforce, efficient data gathering systems and disseminating the final results to all who need to know, not least the participants. The following points may be helpful in structuring the survey design itself:

Review existing data

It is essential to know whether the species has been surveyed before either at local or national level and whether these data are available for comparison.

Choice of survey method

Robustness in both the survey method and planned analytical approach is essential to provide a high level of confidence in the results. Whilst it is essential that the methods deliver the required information there is a balance to be found between the costs and feasibility of more detailed dedicated fieldwork and gathering sufficient information to meet the project aims.

Sampling strategy and allocation

It is often unrealistic to undertake a complete census so a sample survey is best. This sample should be as unbiased as possible, so a random stratified survey is usually chosen.

Recording data – paper or digital or both?

It is now usual to record information online and setting up such a system is a prime consideration for any survey. However, relying on this entirely can be a risk and it is good practice to keep duplicate records as a means of checking data validity - as well as improving data security in the unlikely event the online system fails. Copies can be digital (provided back-ups are kept) or in paper form.

Plan for analysis

It is vital that the data collected is in a format suitable to answer the original project questions and that an appropriate method of analysis is chosen at the outset.

Sharing results – not just how, but why?

The project results must be made available to key decision makers and those involved in undertaking the survey as quickly as possible. This can be through the means of formal publication, online summaries, articles for local websites, newsletters etc. Not forgetting the fieldworkers involved in data gathering! Early availability of results ensures quick follow-up action if required, enables future surveys to be undertaken and inspires confidence and increases motivation amongst the participants.

Case study: The WOS 2021-22 Rook survey

Although WOS had never previously organised a national survey, in 2019 the Society was approached by RSPB, which was undertaking a UK-wide survey of Willow Tit *Poecile montanus*, requesting coordination from County Recorders and additional help from WOS. Since most County Recorders have no track record of volunteer survey coordination (unless they also happen to be a BTO Regional Representative), WOS recognised that uptake of the survey in Wales risked being exceedingly low without further action. WOS, therefore, provided additional coordination and engagement – resulting in good levels of survey coverage.

This encouraging experience, alongside the publication of an updated Birds of Conservation Concern Wales 4 (Johnstone *et al.* 2022) inclined WOS towards a collaborative, partnership-based project to survey a species listed as of particular conservation concern in Wales – one either declining or data-deficient. WOS would take the lead on project organisation and coordination, learning from the experience of the BTO and others. The project would raise the profile of WOS and help to improve links between WOS and the regional network of bird clubs and County Bird Recorders throughout Wales.

Choosing Rook for a single-species survey

Thorpe and Brenchley (2017) identified 25 species as priorities for survey based on population size, conservation status or lack of recent population data. Merlin *Falco columbarius*, Golden Plover *Pluvialis*

apricaria, Dunlin *Calidris alpina*, Short-eared Owl *Asio flammeus* and some seabirds (Gannet *Morus bassanus*, Kittiwake *Rissa tridactyla* and Manx Shearwater *Puffinus puffinus*) require specialist survey fieldwork skills and are more appropriate for professional survey than for volunteers. Some others, including Marsh Tit *Poecile palustris*, Willow Tit, Tree Sparrow *Passer montanus* and Hawfinch *Coccothraustes coccothraustes* have such restricted ranges or small populations that national survey effort is unlikely to add to existing knowledge. The remaining candidates included some woodland specialists (Pied Flycatcher *Ficedula hypoleuca*, Wood Warbler *Phylloscopus sibilatrix* and Redstart *Phoenicurus phoenicurus*), Whinchat, Water Rail *Rallus aquaticus* and Kestrel *Falco tinnunculus*. Willow Tit has since been the subject of a special survey (Wotton *et al.* 2023).

However, it was not until 2022, when the most recent Birds of Conservation Concern report (BoCC4; Johnstone *et al.* 2022) was published, that the plight of the Rook came to the fore. There had been a major change in the conservation status of Rook, which moved from Green to Red as a result of population decline (measured by BBS) of over 50% in the past 25 years.

Survey by volunteers or professionals – things to consider

The Rook is a relatively easy bird to identify and nests colonially in treetops, in groups of nests called ‘rookeries’. Nests are mostly built before leaf burst and so breeding birds are fairly easy to locate. Rooks are still relatively widespread across lowland Wales so this species is very amenable to an all-Wales



A highly visible rookery before leaf burst: Annie Haycock

survey. This survey lends itself to the involvement of a large number of volunteers. Mobility would not necessarily be a barrier to taking part as many rookeries can be located from a car using binoculars. This would be on the proviso that all of the tetrad was visible from roads or public tracks (which is often the case). The majority of rookeries are located below 400m (most below 300m) so survey over much of treeless, upland Wales was unlikely to be needed. Several County Recorders concurred with this general observation of rookery locations with only a handful of records showing where rookeries have traditionally been located at higher elevations.

Engagement and up-skilling has benefits too

WOS thought that birdwatchers and many more general naturalists would be keen to participate in a project that was relatively uncomplicated and provided a general introduction to structured surveys. Such recorders might then be encouraged to take part in further surveys once their confidence and ability had been enhanced by participation in this single-species survey.

Support and funding

This survey was of broad interest to NRW, RSPB Cymru and BTO Cymru; and these organisations came on board to help with the planning, implementation and data analysis stages. Online data collection facilities came from Cofnod (Local Environmental Record Centre, LERC, for North Wales), which, on behalf of LERCs Wales, was the only partner able to provide a system accessible across the whole of Wales. Pembrokeshire Coast National Park provided Ordnance Survey based digital tetrad maps of survey areas. Whilst some of this support was provided for free, the online booking system and the detailed data analysis involving modelling and population estimates was funded by NRW.

The planning phase

The scientific objectives of the 2022/23 survey can be summarised as follows:

- To produce a population estimate for Wales comparable with the 1996 UK survey, and potentially with that for 1975.
- To produce population estimates and distribution maps for the WOS recording areas (equating to the Watsonian vice counties).
- To calculate average rookery size for the whole of Wales and for each recording area.
- To identify the tree species chosen for nesting.

As mentioned before, another aim of the project was to engage with a large number of volunteer surveyors, preferably many who had not taken part in a survey before and might, therefore, be encouraged to engage with other surveys in the future.

The author agreed to be the overall coordinator and, as it was vital to gain their support, to further provide liaison with BTO Cymru, RSPB Cymru, County Bird Recorders, Affiliate groups and other bird clubs and societies, in order to find volunteer surveyors. The WOS Communications Group coordinated survey promotion, with support from partner organisations.

The appointment of Local Organisers, who had both local geographic knowledge and local contacts, was the most efficient means of ensuring local participation. BTO Regional Representatives (RRs) were particularly well positioned to do this work as most of them were already experienced in coordinating BTO surveys and had contacts with local volunteers who could be approached to take part. Although this was not a BTO survey, it did have BTO endorsement and most RRs agreed to help. Local Organisers, with the exception of Radnorshire, were appointed by the end of 2021.

Choice of survey method

In so far as possible, the aim was to repeat the method adopted by the 1996 UK survey (Marchant and

Gregory 1999). The organisers of similar concurrent surveys planned in England (e.g. Cornwall and Somerset) recommended that the survey be extended over two consecutive years. This would both maximise effort and help provide sufficient data for the calculation of a robust population estimate.



A pair of Rooks at the nest: John Hawkins

In 2022, the survey requested observers to count the number of nests in rookeries on a single visit between 1 March and 15 April, before the leaves fully emerge. In the case of a late spring, the survey could be extended to 30 April. In 2023, the recommendation was to begin nest counting no earlier than mid-March and to extend it to the end of April. This difference was based on feedback from Local Organisers, County Recorders and information from Griffin (1999) that rookeries did not reach their peak size until later in the nesting season and early counts would result in a sub-maximal count. The definition of a rookery was two or more nests in close proximity but over 100m from another cluster of nests. Where possible the tree species in which the rookery was located were identified.

Methods were summarised in a guidance document, produced with input from key Local Organisers and provided to volunteers. This was particularly helpful to volunteers new to bird surveys.

Review existing data

The BTO provided data, at tetrad level, from the results of the following historic surveys:

1. 1975 rookery survey
2. 1980 sample survey
3. 1996 rookery data – same scale as 1975

4. 2007-11 Bird Atlas data at tetrad level

Sampling strategy and allocation

The core survey was a random stratified sample of tetrads covered in previous surveys. Based on advice from Simon Wotton, RSPB Centre for Conservation Science, three strata were proposed:

1. All tetrads surveyed in 1996 (n=173), where direct comparison between surveys was possible;
2. An additional random sample of 15-20% of the 1,074 tetrads surveyed in 1975/80 (n=161 to 215), where methods were similar but not perfectly comparable;
3. A random sample of 10-15% of the remaining 1,174 tetrads occupied in the 2007-11 Bird Atlas (n=117 to 176), indicating where breeding Rooks were likely to be encountered but the methods were not comparable.

NRW produced the list of random stratified tetrads to be surveyed in 2022. This Priority list contained all 173 tetrads surveyed in 1996 and these were designated as Stratum One. The 221 tetrads from the 1975/80 surveys were designated as Stratum Two. The Stratum Three category included 175 tetrads from the 2007-11 Bird Atlas (Balmer *et al.* 2013). This resulted in a total of 568 Priority tetrads for survey in 2022. This number was subsequently reduced to 562 when ground-truthing revealed that six tetrads were either wholly outside Wales (in England) or located in the sea. These 562 were allocated to Local Organisers. As most Local Organisers were BTO Regional Representatives (RR's) it made sense to use the BTO regions as the main data collection/recording areas. They are based on 10-km squares and broadly equate to Watsonian Vice-Counties although there is some variation around region boundaries. Data were assigned to WOS Recording Areas after data collection.

Delivery of the Rook Survey

Survey coverage

At the end of the 2022 season 90% of Priority tetrads had been booked. Inevitably not all of these bookings translated into completed surveys but data were entered online for 79% (450 Priority tetrads). Due to the success of the 2022 season, at the end of the year NRW released a second list of random stratified tetrads, minus those Priority tetrads already surveyed in 2022. At the end of the 2023 field season the total number of Priority tetrads with data entered online was 741. Records for a further 286 additional (Extra) tetrads (Table 1) and many casual records were also entered resulting in 2,356 unique records online.

A few surveyors specifically indicated, in the online comments section, that not all the tetrad had been surveyed but in general an assumption was made that the whole area within the priority tetrads was surveyed for the presence of rookeries.

The Coordinator and Local Organisers did not always have sufficient local knowledge to identify and remove duplicate records. For example, Haycock and Haycock (2024) in their Pembrokeshire Rook Survey report identified several examples where different observers gave different site names and 6-fig. grid references for what was known to be the same rookery. 62 tetrads, in the Wales survey, were considered incomplete or were flagged and subsequently discounted from analysis due to unresolved anomalies in data entry. This left records from 679 tetrads that could be included in the analysis (Hereward *et al.* 2024).

At least 350 volunteers took part in the survey. This number represents the names entered online; additional surveyors were known to have submitted data to Local Organisers who entered data on their behalf (but did not always provide the name of the surveyor). Several volunteers, new to surveying, provided feedback that they would be interested in taking part in future surveys indicating that part of the overall aims of the project may have been achieved. However, there is no evidence to date to show that this has happened.

Table 1. The number of priority and extra tetrads surveyed in each WOS recording area. Those with rookeries and those without.

WOS recording area	Tetrads surveyed						Totals of all tetrads
	Priority, with rookeries	Priority, with no rookeries	Number of priority tetrads	Extra, non-priority, with rookeries	Extra, non-priority, with no rookeries	Number of extra tetrads	
Anglesey	19	20	39	1	0	1	40
Brecon	18	52	70	15	3	18	88
Caernarfon	16	38	54	22	25	47	101
Carmarthen	13	37	50	1	3	4	54
Ceredigion	20	34	54	12	2	14	68
Denbigh	35	56	91	8	5	13	104
East Glamorgan	11	37	48	33	36	69	117
Flint	14	22	36	17	14	31	67
Gower	7	20	27	20	8	28	55
Gwent	18	43	61	2	1	3	64
Meirionnydd	7	19	26	0	0	0	26
Montgomery	29	34	63	5	0	5	68
Pembroke	35	55	90	39	7	46	136
Radnor	8	24	32	4	3	7	39
TOTALS	250	491	741	179	107	286	1027

Coverage was fairly uniform across Wales with no region having noticeable gaps (Fig. 1). Of the 171 tetrads in Stratum One originally chosen for survey, 146 were covered and of the remaining uncovered squares, 9 were in Carmarthenshire, where take-up of the survey was generally poorer than in other regions. Of the remaining priority tetrads (Stratum Two and Three), a further 263 and 270 tetrads respectively were fully complete with records entered for a further 62 tetrads where it was unclear whether a full tetrad survey had been undertaken. In 2023, take-up of the survey was fairly uniform with gaps in Ceredigion, Carmarthenshire and Gwent. However, none of these gaps in coverage were significant.

Rookeries were found at low elevations, mostly below 250m and very rarely above 400m. Of the priority tetrads, approximately one third (33.7%) were occupied. This was higher than the 24% found in 1996 (Marchant and Gregory 1999). Tetrad occupancy varied from 22.9% in East Glamorgan to 46.0% in Montgomeryshire.

Data checking phase

Once the survey had begun and some data entered online, the coordinator and the Local Organisers began checking the validity of the data and dealing with queries. Common errors included grid references that did not match the tetrad number, or data fields not completed.

Data overview supplied to analysis

In total, 1,205 individual rookeries were counted, which together held 26,762 nests (Table 2 on p.22). This survey assumed that all nests counted were occupied. A very few surveyors did take time to evaluate nest occupancy, leading to a slightly reduced count compared with visible nests but this was not a pre-requisite of the survey and it was considered unlikely to significantly alter the overall results.

The uptake of additional survey tetrads was variable across the regions. Pembrokeshire undertook the

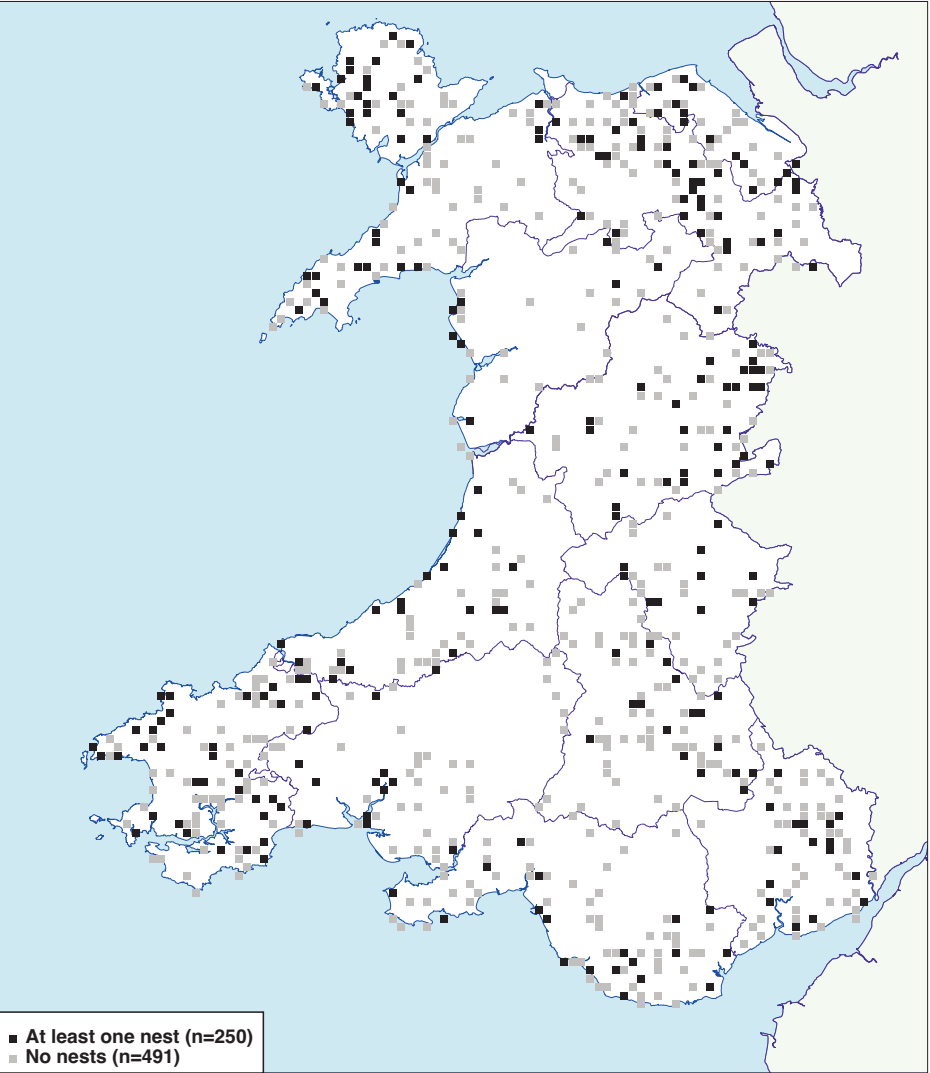


Figure 1. Map showing the distribution of Priority tetrads surveyed 2022/23 (black squares contained rookeries that were counted; no rookery was found in grey squares).

most complete survey and the results (Haycock and Haycock 2024) provided more detail than the national survey could achieve. East Glamorgan and Gower also undertook many additional tetrads, primarily covering those where rookeries were known to be present during the 2007-11 Bird Atlas period.

WOS recording area	PRIORITY TETRADS		EXTRA TETRADS		CASUAL RECORDS	
	No. of rookeries	No. of nests	No. of rookeries	No. of nests	No. of rookeries	No. of nests
Anglesey	33	392	1	36	44	689
Brecon	29	746	22	623	24	402
Caernarfon	30	432	32	639	16	312
Carmarthen	17	393	1	5	63	1392
Ceredigion	37	845	20	775	30	775
Denbigh	72	1511	8	123	51	942
East Glamorgan	24	335	45	1035	4	58
Flint	22	601	26	445	16	207
Gower	10	173	50	749	1	8
Gwent	25	387	3	154	44	741
Meirionnydd	13	384	0	0	34	900
Montgomery	63	1153	6	134	51	1332
Pembroke	51	1971	62	2268	77	2060
Radnor	12	225	9	142	27	278
TOTALS	438	9548	285	7128	482	10096

Table 2. Number of rookeries and total number of nests recorded in each WOS recording area from priority tetrads, extra tetrads and all casual records not from surveyed tetrads.

Discussion of field results

Rookery size

Average rookery size across Wales during the survey was 23.1 nests, which represents a decrease from 25.4 recorded in 1996 and 25.2 in 1975. The majority of rookeries were small, less than 20 nests (Fig. 2). Rookeries in excess of 50 nests were far less frequent than in previous surveys.

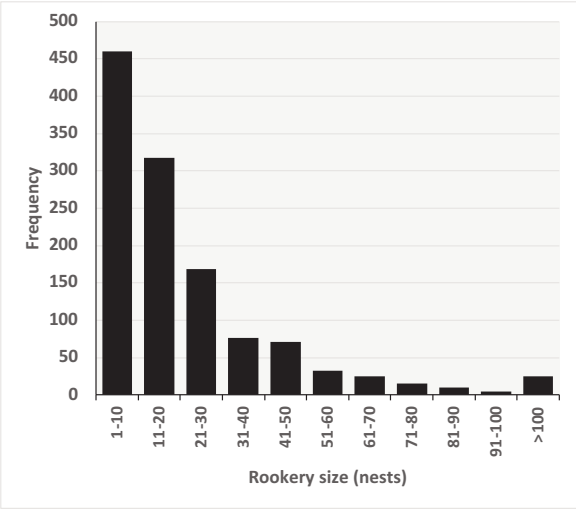


Figure 2. The frequency distribution of rookery size across Wales using all records with counts (Priority and Extra tetrads) and Casual records.

Pritchard *et al.* (2021) reported that no rookery count exceeding 200 nests had been published since

1996. This survey found three rookeries exceeding 200 nests, all located in Pembrokeshire. A fourth colony, counted under this survey, was located just over the border in England (Cheshire). Only 21 rookeries exceeding 100 nests were found during this survey, most of which were located in Pembrokeshire. This finding was much the same as the data from 1996, and the nest count for the largest rookery in each WOS recording area is given in Table 3.

Since 1996, rookery size appears to have decreased across Wales and this is also apparent in the detailed local analysis from Pembrokeshire (Haycock and Haycock 2024). Rookery size changes between 2000 and 2017 were less marked in Gower, where annual rookery counts are made (*e.g.* Tallack 1997). Prior to 1996, average rookery size was relatively stable and a detailed survey in Ceredigion (Chater 1996), where rookery size has always been higher than the national average, highlights this.

Table 3. The average rookery size and the largest rookery count in each WOS recording area using records from all priority tetrads, Extra tetrads and Casual records.

WOS recording area	Numbers of nests	Numbers of rookeries	Average rookery size (nests)	Largest rookery (nests)
Anglesey	1117	78	14.3	55
Brecon	1771	75	23.6	105
Caernarfon	1383	78	17.7	115
Carmarthen	1790	81	22.1	71
Ceredigion	2395	87	27.5	171
Denbigh	2576	131	19.7	116
East Glamorgan	1428	73	19.6	76
Flint	1253	64	19.6	200
Gower	930	61	15.2	47
Gwent	1282	72	17.8	64
Meirionnydd	1284	47	27.3	133
Montgomery	3437	108	31.8	168
Pembroke	6650	208	32.0	222
Radnor	645	48	13.4	62
TOTAL	27941	1211	23.1	

Spatial distribution of rookeries

In the tetrads where rookeries were found, 62% contained only a single rookery. Very few tetrads held three or more rookeries. Clearly this statistic depends on the observers following the strict definition of a rookery but is most likely to be related to the food availability in the area surrounding each rookery (Brenchley 1986).

WOS Recording Area	Tetrad	6-fig grid ref.	Location Name	Rookery size (no. nests)
Pembroke	SM93C	SM901356	St Nicholas	222
Pembroke	SM72M	SM756253	St David's group	215
Pembroke	SM72X	SM794245	Llanunwas	202
Flint	SJ36R	SJ371622	*Balderton Village, Welsh Road	200
Ceredigion	SN57T	SN579764	Chancery (Conrah Hotel)	171
Montgomery	SJ20A	SJ208000	Nr Refail	168
Pembroke	SM81S	SM864143	Haroldston woods (south end)	138
Meirionnydd	SJ04G	SJ039435	Ddwryryd (Druid) just west of A55/A494 junction	133
Pembroke	SM83Q	SM879313	Mabws Bridge/caravan park, near Mathry	127
Meirionnydd	SH93E	SH913382	Pont Ty'n-y-ddol, Frongoch, Bala	118
Denbigh	SJ15L	SJ151523	Nr Llysfas College, Pentre Celyn	116
Caernarfon	SH12U	SH164290	Carreg, Aberdaron	115
Meirionnydd	SJ03P	SJ042385	Hendwr Caravan park drive, Llandrillo	115
Caernarfon	SH33H	SH334342	Penrhos Polish Village	113
Pembroke	SM83L	SM842324	Trefin	112
Ceredigion	SN34A	SN305405	Newcastle Emlyn	111
Pembroke	SM93M	SM944341	Scleddau	107
Meirionnydd	SJ04G	SJ039435	NW of Druids Crossroads	107
Pembroke	SN04Q	SN067402	Near Nevern	105
Brecon	SO03I	SO031371	Bwlch-henllan Farm, Lower Chapel	105
Pembroke	SM91U	SM962190	Withybush airfield	105

Table 4. The largest rookeries from all records (priority and extra tetrads and casual). *The Balderton village rookery was within a surveyed tetrad but was actually on the English side of the boundary.

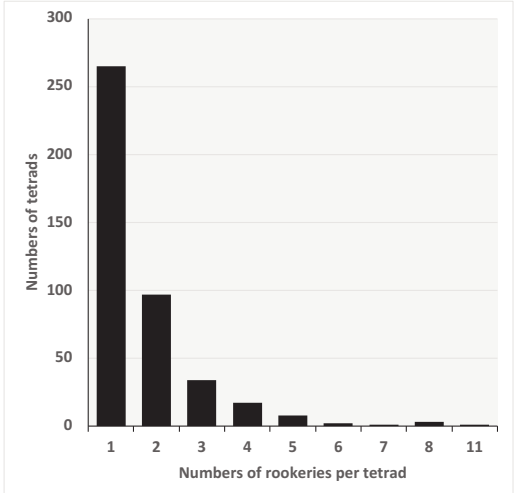


Figure 3. Frequency distribution of rookeries per surveyed tetrad (priority and extra n=429) where rookeries were present.

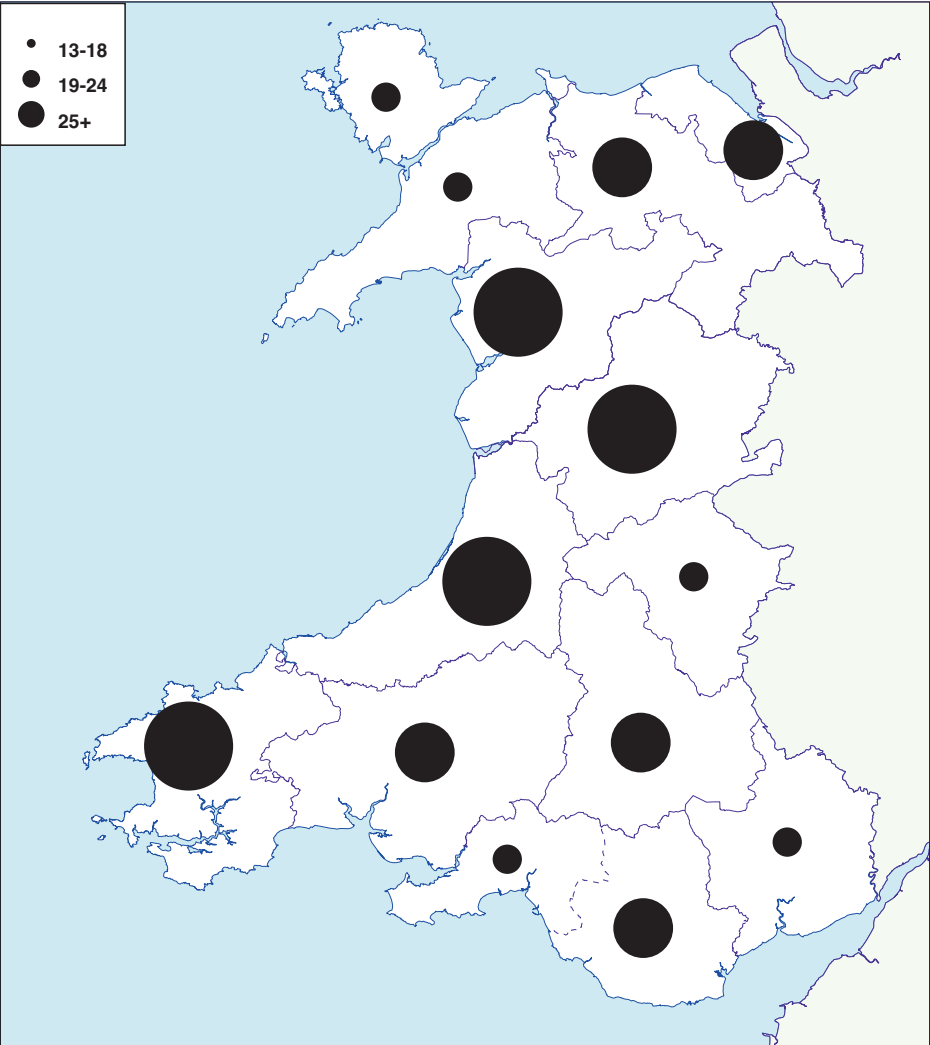


Figure 4. Average rookery size per WOS recording area using all rookery count data (priority and extra tetrad and casual records).

Tree species in which rookeries were recorded

Identifying tree species can be challenging, particularly before leaf-burst. 29 rookeries had no tree species category identified but from the remaining rookeries where a broad category was given, nests were predominantly located in broadleaved trees (82.1%). 11.3% of rookeries were located in a mix of broadleaves and conifers - although where more details were given, even here broadleaved trees predominated. Only 4.3% of rookeries were located entirely in conifers (Table 5). This pattern of tree species use mirrored that from previous surveys. There were some regional differences and Pembrokeshire in particular seemed to have a greater proportion of rookeries located in a mixture of broadleaves and conifers, or in conifers only.

WOS Recording Area	Priority Tetrads			Additional Tetrads			Casual Records		
	B/L	MXD	CON	B/L	MXD	CON	B/L	MXD	CON
Anglesey	32	1	0	1	0	0	34	3	0
Brecon	21	4	3	19	2	1	22	2	1
Caernarfon	27	3	0	27	2	3	13	3	0
Carmarthen	14	2	1	1	0	0	55	6	2
Ceredigion	24	4	4	16	5	0	16	5	2
Denbigh	61	8	1	8	0	0	45	5	1
East Glamorgan	21	1	2	36	8	1	3	1	0
Flint	19	2	0	24	1	1	12	1	0
Gower	6	1	3	27	12	11	0	0	1
Gwent	19	3	1	3	0	0	40	2	1
Meirionnydd	7	4	2	0	0	0	25	6	1
Montgomery	56	5	1	5	0	0	48	3	0
Pembroke	54	9	6	74	13	2	88	13	2
Radnor	12	0	0	8	0	1	20	6	1
TOTAL	373	45	24	249	43	19	421	56	12

Table 5. The number of rookeries identified as being in Broadleaved trees (B/L), a mixture of broadleaves and conifers (MXD) and conifers only (CON).

The broadleaved species used broadly mirrored the most common tree species available, notably Oak *Quercus* spp., Ash *Fraxinus excelsior*, Beech *Fagus sylvaticus* and Sycamore *Acer pseudoplatanus*. However, a wide range of other species were also recorded: Alder *Alnus glutinosa*, Birch *Betula* spp., Elm *Ulmus* spp., Eucalyptus *Eucalyptus* spp., False Acacia *Robinia pseudoacacia*, Hazel *Corylus avellana*, Holly *Ilex aquifolium*, Horse Chestnut *Aesculus hippocastanum*, Lime *Tilia* spp., Poplar *Populus* spp. and Willow *Salix* spp. The conifers identified were predominantly Scots Pine *Pinus sylvestris* with occasional mention of Corsican Pine *Pinus nigra*, Monterey Pine *Pinus radiata*, Larch *Larix decidua*, Lawson Cypress *Chamaecyparis lawsoniana*, Spruce *Picea* spp., and Monkey Puzzle (Chilean Pine *Araucaria araucana*).

Discussion of Rook conservation status

The fact that a species so well-known - and once so common that it was regarded as an agricultural pest - has now declined significantly, is of deep concern. The reasons for this decline are likely to be due to a combination of factors and further research may be necessary to quantify the major drivers of change at different spatial- and temporal scales. One large-scale pressure known to impact Rook is agricultural change. The major changes in agriculture since the Second World War have been well documented (Burns *et al.* 2016). Whilst major land-use changes were most evident in the immediate post-war period, farming practices have introduced more subtle changes in recent years. These latter changes, including agrochemical regimes, sward and grass ley management, are likely to have influenced the abundance and availability of soil invertebrates, which are key sources of food for Rooks.

Decreasing rookery size may increase the species' vulnerability to external pressures. For example, smaller rookeries (and those with more dispersed nests) may be more vulnerable to aerial predation. Anecdotal observations of nest predation by Raven *Corvus corax*, Goshawk *Accipiter gentilis* (Pete Jennings pers. comm.), and Red Kite *Milvus milvus* - Fred Slater (pers. comm.) were provided during the period of this 2022/23 survey. Rooks exhibit significant nest-defence behaviour when rookeries are approached by potential aerial predators (Röell and Bossema, 1982). Larger and more compact rookeries are more likely to be successfully defended than those that are small or where nests are more dispersed.

Secondary pressures on Rook populations may arise from changes in the distribution or abundance of favoured tree species. Traditionally some tree species have been particularly selected for nesting, presumably for structural reasons. Elms were once a favoured species, but Rooks were forced to change location after Dutch Elm Disease *Ophiostoma novo-ulmi* significantly reduced the number and distribution of *Ulmus* spp. in the 1970s. Ash became one of the more favoured species along with Oak, Beech and Sycamore; and another recent disease outbreak (Ash die-back *Hymenoscyphus fraxinus*) has the potential to change this species' availability across Wales. Significant loss of Ash in the landscape could lead to many traditional rookery locations being abandoned. Oak, Beech and Sycamore are still common trees on Welsh farmland so it is unlikely that nesting sites will be a direct limiting factor in population decline, but rookeries may fragment and then become vulnerable to other ecological pressures.

Historically, Rooks have been subject to control and persecution. In this survey, several former rookeries were reported by surveyors as having been abandoned as a result of human activity in recent years. Since Rook was removed from the General Licence in Wales in 2023, the species can only be controlled by the grant of a specific licence, which should reduce the likelihood of this impact at large spatial scales. Active rookeries can be noisy and produce a lot of droppings beneath the trees, and incidences of active (non-lethal) disturbance have been noted in the past. An additional direct disturbance to rookeries may come from tree-felling. Rookeries are commonly located in roadside trees and such trees may be felled for reasons of safety. Several local authorities have taken a precautionary approach to Ash die-back, and felled trees (even those not yet showing signs of disease) alongside major roads. There are a range of other activities that may trigger tree-felling, including developments, building safety, or other tree diseases.

A final possible pressure on rookeries is the changing weather patterns associated with climate change. Increasing storm activity due to changing weather patterns may result in tree damage and/or nest loss or damage during the breeding season. It seems likely that this may have an impact on some rookery locations in recent years.

Lessons learned from the WOS Rook survey

A number of important lessons have been learned through the design and delivery of the WOS Rook Survey, some more relevant to future Rook surveys building on the national baseline data from the present survey, and others applicable to any future single-species volunteer survey coordinated by an organisation like WOS.

Lessons applicable for future Rook surveys

Survey design: spatial resolution and the definition of a rookery

The strict definition of a rookery, 'any active Rook nest or group of nests, 100m or more from the next nearest nest' was introduced in the 1975 survey (Sage and Vernon 1978). Applying this definition proved more difficult than the definition itself might seem to imply. Many of the larger rookeries were not simple clumps of nests, but often groups of nests more-or-less within 100m of each other. These 'aggregates' or groups with satellites were particularly evident in Pembrokeshire. Linear rookeries also proved to be another challenge for site identification.

Identifying duplicate records also proved a challenge and the accuracy of the grid reference supplied was important. In theory, if a six-figure grid reference differed, even by a single figure, from another, that should indicate a different rookery. In reality, grid references estimated from paper maps by different observers for the same rookery might differ from each other by this much, an error compounded by the distance from which the rookery was viewed. Requiring eight-figure grid references would go a long way towards eliminating this problem and reduce the generation of duplicate records.



A linear rookery: Annie Haycock

For linear rookeries, observers should be asked to record the middle of the rookery. In Gower and Pembrokeshire, where rookeries are often counted on an annual basis, each is assigned a standard name to avoid such errors in rookery locations. This is not practical in all regions because it requires a lot of pre-survey work from Local Organisers; it may not be practically possible in areas with insufficient local knowledge.

Survey design: there are advantages to a single-year survey!

Running the survey over two years, whilst adding significantly to the data gathered, did bring many complications. Many survey organisation and data checking issues would have been overcome if the survey had been a single year. However, this would have required a much longer lead-in time at the planning and promotion phase than was possible for this survey.

Survey methodology: leaf-burst and late nesters

The choice of time period in which to undertake the survey was a trade-off between obtaining the most accurate data (maximum nest counts) and allowing surveyors sufficient time in which to do the survey. Nests are best seen before leaf-burst, but many rookeries increase in size as the season develops (Griffin 1999; Haycock and Haycock 2024; Tallack 1997). The timing of leaf-burst is somewhat variable and likely advancing due to factors related to climate change (Vitasse *et al.* 2009). In 2022, survey guidance suggested that rookeries be counted between 1 March to 15 April. In 2023 this was changed to 15 March to end of April, but the majority of the tetrads had already been surveyed. The early count period meant that the number of nests counted might therefore have been less than the true maximum. Future breeding Rook surveys could take these factors into account and consider count timing in the context of contemporary leaf-burst timing.

Lessons applicable to other single-species surveys in Wales

Species ecology and survey design

Despite the fact that Rooks have been subject to well-documented local and national surveys for at least 100 years, and that counting the nests of a colonial species is relatively easy compared to either nests or individuals of many other species, it is always important to re-evaluate the target species' ecology and consider any new influencing factors, e.g. climate change impacts. There is a balance to be found between exact methodological comparability and acceptable modifications that might be expected to increase the accuracy of the results.

Survey organisation

It is essential to plan any future survey very carefully, using as much available expertise and lead-in time as possible to be fairly confident that the project will achieve its aims. The planning and organisation of the Rook survey evolved over time as partners came on board, further local knowledge and expertise became available and data handling systems were created. This was not ideal, and although it was not thought to significantly affect the outcome of the current project, it did increase the survey organisation time.

Lead-in time

The methods for the Rook survey were relatively simple compared to those that might be needed for many other species. Where more complicated data gathering is required, more planning time is needed to produce practical methods and test them in the field in advance of the survey. Planning should begin at least two years in advance of a survey being undertaken in order to ensure many of the teething problems have been identified and resolved long before actual data are gathered.

Critical role(s) of the coordinator

A dedicated project coordinator is critical to the success of the project. This role requires excellent organisation and communication skills, endless patience and commitment to the project outcome. The role is best undertaken by a single individual who acts as the first point of contact and is completely familiar with the project methods, responding to queries promptly and effectively. Experience of working with volunteers is a great advantage.

However, the coordinator is unlikely to have sufficient range of local contacts to be able to contact enough volunteers to undertake such a widespread survey. Local Organisers (LOs) who can directly contact many potential volunteers and also help with advertising and promoting the project, must be recruited. Regular communication between the coordinator and the LOs is then essential to share information and motivate volunteers.

Advertising, promotion and engagement are linked

Widespread awareness of the survey and clear information about how volunteers can take part is vital. Local bird clubs and the existing volunteer workforce is a good basis but new volunteers are required. Contact with conservation groups/wildlife organisations, either directly or through social media, needs to be planned well in advance of the beginning of the survey season. The communications groups within the organising partners play a key role here.

Funding for critical steps

Whilst the bulk of the person-time in a survey project will be contributed by volunteers, whose time is given freely, there are essential project costs that must be established and secured prior to the project starting.

1. Coordinator time
2. Booking system
3. Volunteer expenses
4. Paid fieldwork time to fill in critical gaps
5. Analysis

For the Rook survey, the coordination work was done by volunteers but it should not be assumed that this will be true for any future surveys. Similarly, volunteer expenses were made available but very few claims were received, and no paid survey time was required due to the relative simplicity of the survey methods. None of this should be assumed for future survey projects.

Key field and data checklist

- ✓ Clear aims including analytical plans
- ✓ Clear guidance for LOs and surveyors - including guidance for data input
- ✓ Recording systems and form design may be different for different species
- ✓ Welsh language - Bilingual communications, instructions and guidance are highly desirable.
- ✓ It is vital that surveyors are requested to keep copies of their data so that any queries can be answered promptly.

Data input systems, checking and data retrieval

In the foreseeable future it is likely that the LERCs Online Recording Systems will continue to be used for data input. In the case of the Rook survey, Cofnod, the LERC for North Wales, offered to help as their systems could be adapted to cover all of Wales. Within the capacity of their staff resources, a tailored recording form was created for the Rook survey and this generated spreadsheets available for analysis but there was little time to test out the system in advance of the survey season. Future surveys will require more time for system design and checking before going live. Where possible the intended data analysts (or at least an analyst with relevant experience) should be involved in testing the system and influence the format and structure of the data outputs.

After the survey is complete

Communications and feedback of survey delivery and results (including analytical results) is particularly important to funders, partners and contributors as well as the volunteer surveyors themselves. The type and timing of communications and feedback needs to be carefully planned, and certainly not left as an afterthought.

Assuming all these points are carefully considered and taken into account, it is quite likely that WOS, along with partners, will be undertaking further single species surveys in the years to come

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Status and distribution of Rook *Corvus frugilegus* in Wales 2022/23

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Abstract
The Rook *Corvus frugilegus* is a well-recognised colonial breeding species associated with farmed landscapes. Over the last century, Rook specific surveys identified fluctuating populations. The most recent UK survey was in 1996. In the latest BoCC review, the species’ conservation status in Wales was upgraded from green to red. In 2022 and 2023 the Welsh Ornithological Society (WOS) coordinated a countrywide colony survey. Coverage was based on tetrad searches and colony counts, following a stratified random design. To estimate population and distribution size and change, a Random Forest modelling approach was implemented for both 1996 and 2022/23 datasets, using a two-step approach modelling presence/absence of Rook colonies in each tetrad, followed by the abundance of Rook nests (colony size) where present. These model predictions were applied to all tetrads in Wales and a bootstrapping approach taken to estimate total Rook nest abundance at each spatial scale. Differences in distribution, abundance and change in the average number of breeding pairs per rookery were tested for significance using generalized linear models (GLMs). The current Welsh Rook breeding population was estimated to be 44,127 pairs (95% confidence limits (CI): 41,144-46,940). This was estimated to represent a decline in the abundance of Welsh Rook breeding pairs nationally and a 5.6% decline in the number of occupied tetrads in Wales since 1996. The average number of breeding pairs per rookery (size of rookeries) has also declined between 1996 and 2022/23, from 25 nests in 1996 (CI: 22–28) to 20 nests in 2022/23 (CI: 19–21), a decline of 20%. There was substantial regional variation in both distribution and abundance change. The drivers of Rook declines in Wales are unclear and will require further investigation if the deteriorating conservation status of Rook in Wales is to be addressed.

Crynodeb
Mae’r Ydfran *Corvus frugilegus* yn rhywogaeth sy’n nythu’n gytrefol ac a gysylltir â thir amaethyddol. Yn ystod y ganrif ddiwethaf, dangosodd arolygon arbennig o’r Ydfran amrywiadau ym maint y boblogaeth. Cynhaliwyd yr arolwg diwethaf yn y DU yn 1996. Yn y fersiwn diweddaraf o’r AoBC, newidiwyd statws cadwraeth y rhywogaeth yng Nghymru o wyrdd i goch. Yn 2022 a 2023, trefnodd Cymdeithas Adaryddol Cymru (CAC) arolwg cenedlaethol o nythfeydd. Seiliwyd yr arolwg ar chwilio am nythod ymhob tetrad a chyfrif nythfeydd, yn dilyn cynllun hapsampl haenedig. Er mwyn amcangyfrif maint y boblogaeth a maint ac amrywiad dosbarthiad, defnyddiwyd modelu Random Forest ar gyfer setiau data 1996 a 2022/23, gan ddefnyddio dull dau gam, yn modelu presenoldeb/absenoldeb nythfeydd Ydfran ymhob tetrad, ac yna nifer y nythod Ydfran (maint y nythfa) lle’r oedd rhai’n bresennol. Defnyddiwyd y daroganau model hyn ar gyfer pob tetrad yng Nghymru a defnyddiwyd dull ymlwytho i amcangyfrif y nifer o nythod Ydfran ar bob graddfa ofodol. Profwyd gwahaniaethau mewn dosbarthiad, niferoedd a newidiadau yn y nifer o barau ar gyfartaledd ymhob nythfa am arwyddocâd trwy ddefnyddio

modelau llinellol cyffredinodedig (GLM). Amcangyfrifwyd fod y boblogaeth Ydfran sy’n nythu yng Nghymru yn 44,127 pâr (Ch 95%: 41,144-46,940). Roedd hyn yn dangos gostyngiad yn y nifer o barau Ydfran yn nythu yng Nghymru, a gostyngiad o 5.6% yn y nifer o detradau oedd yn cynnwys nythod Ydfran ers 1996. Bu gostyngiad hefyd yng nghyfartaledd y nifer o barau o fewn nythfa (maint y nythfeydd) rhwng 1996 a 2022/23, o 25 nyth yn 1996 (Ch: 22–28) i 20 nyth yn 2022/23 (Ch: 19–21), gostyngiad o 20%. Roedd cryn amrywiaeth yn y newidiadau mewn dosbarthiad a niferoedd rhwng gwahanol ranbarthau. Nid yw’r rhesymau am ostyngiad yn nifer yr Ydfran yng Nghymru yn eglur, a bydd angen ymchwil pellach os am gymryd camau i wella statws cadwraeth yr Ydfran yng Nghymru.

1. Introduction
1.1. Rook population change

The Rook *Corvus frugilegus* is a well-recognised and colonial breeding species of Welsh and UK farmed habitats (Holyoak 1972; Feare 1978; Marchant and Gregory 1999). Over the last century, fluctuations in the Rook population have been documented in a series of UK-wide surveys. Between the 1944-45 and 1975 survey declines were identified at UK and Welsh scales (UK: -43% and Wales: -60%), with a subsequent increase between the 1975 and 1996 surveys (UK: +39%, Wales: +36%; BTO 1947; Sage and Vernon 1978; Brenchley 1986; Marchant and Gregory 1999). More recently, the Breeding Bird Survey (BBS) trend highlights a return to population decline (Heywood *et al.* 2024) and Rook has been moved from Least Concern to Vulnerable on the IUCN European Red List (BirdLife International 2023). In Wales, the recent decline has been particularly marked, with the Breeding Bird Survey (BBS) index for Wales falling by 59% between 1995–2022, accelerating in the last decade to suggest a 45% decline (2012-2022). This rate of decline is greater than in any other UK country (Heywood *et al.* 2024). In consequence, the Rook’s conservation status was moved directly from the Green list to the Red list in the most recent Welsh assessment of Birds of Conservation Concern (BoCCW4; Johnstone *et al.* 2023).

The last UK-wide Rook survey was performed in 1996. Given the time since this survey and concerns about population decline and conservation status, the Welsh Ornithological Society (WOS) coordinated an all-Wales Rook survey, in partnership with the British Trust for Ornithology (BTO), Cofnod, Royal Society for the Protection of Birds (RSPB) and Natural Resources Wales (NRW). The survey was conducted by volunteer fieldworkers in the breeding seasons of 2022 and 2023. This paper presents the results of population and spatial analysis of the new survey, and a comparison with remodelled results using the data from the 1996 survey in order to understand and refine our estimates of change in conservation status.

1.2. Aims
BTO was commissioned by Natural Resources Wales (NRW) to conduct bespoke analysis of the 2022 - 2023 Welsh Rook survey data (henceforth referred to as the ‘2022/23 survey’) in order to produce contemporary population estimates and distribution maps, and to compare these with data from the 1996 UK Rook survey.

The aims of the survey and subsequent analysis were to:

- estimate the population size of Rooks at three spatial scales (Wales, NRW Area Statement areas, WOS Recording Areas);
- assess the spatial distribution of Rooks in Wales;
- recalculate the population size and distribution of Rooks in Wales in 1996, and assess changes in abundance and distribution since 1996;
- consider whether the average size of rookeries (based on colony counts of Apparently Occupied Nests, AON) has changed since the 1996 survey.

2. Methods

Survey design and coverage

The 2022/23 survey was undertaken over two breeding seasons (2022 and 2023). Fieldwork was entirely undertaken by volunteers, co-ordinated at Wales national scale by WOS (<https://birds.wales/rook-survey/>) and organised at vice-county level by BTO Regional Representatives (RRs) and County Bird Recorders. The survey method was a tetrad (2 km x 2 km) search with counts of found colonies. Tetrad selection followed a stratified random design across the known range of Rook in Wales and representative of counties, enabling population estimates to be produced at the required spatial scales. Volunteers were also encouraged to survey additional tetrads to the stratified random sample, at their own discretion.

Distribution and population estimates were required for Wales, for each NRW Area Statement area (Appendix 1 Figure 1a) and for WOS Recording Areas (<https://birds.wales/counties-map/>; Appendix 1 Figure 1b) [all the appendices are available online at: <https://birds.wales/wp-content/uploads/2024/10/Hereward-et-al.-RookReport.-Milvus.-NEW-Appendicesv2-1.pdf>]. Note that NRW Area Statements outline the key challenges (or themes) facing a particular Welsh spatial area (broadly defined around catchments), and the actions that can be taken to remedy these (<https://naturalresources.wales/areastatements?lang=en>). WOS Recording Areas are long-term historic regional recording boundaries, many of which equate to vice-counties (VC) but with boundary amendments in some cases; of particular note is VC41 - Glamorganshire, which is divided into East Glamorgan and Gower WOS Recording Areas (Pritchard *et al.* 2021). Survey tetrad allocation was supported by BTO and data collated by Cofnod (<https://www.lercwales.org.uk>) on behalf of LERC Wales.

Field methods

The 2022/23 survey followed the same methodology as for the 1996 survey, detailed in Marchant and Gregory (1999). Surveyors were asked to survey each selected tetrad once before full spring leaf emergence (typically between 1st March – 15th April, although up to the 30th April was accepted in case of a late spring). The surveyors were asked to find all rookeries within the tetrad and count the total number of nests per rookery (Appendix 2). A rookery was defined as two or more nests in close proximity to each other, which were over 100m from any other cluster of nests. Where a rookery spanned multiple tetrads, all the nests in the rookery were recorded as being located within the tetrad with the most nests in it. Note that, as was the case in Marchant and Gregory (1999), the count of the number of active nests (AON) is assumed to equate to the number of breeding pairs.

Data processing

Only data from selected sample tetrads were analysed, with one exception. The general inclusion of additional data recorded from tetrads outside of the stratified random sample could have skewed model outcomes, especially if the opportunistically-surveyed tetrads were biased towards known large rookeries. However, Random Forest models cannot predict beyond the range of observed data, and after initial testing of the 2022/23 model predictions (see below), we noted a tendency to underpredict spatial clusters containing larger rookeries. To overcome this, we added the single opportunistically-sampled tetrad with the largest count of Rook nests to the stratified random sample, in order to expand the range of observed counts in the training dataset for modelling. This substantially improved model performance.

Data were further processed to remove duplicates and, where necessary, to improve information on survey counts or coverage. Duplicate surveys of the same tetrad were identified, and the visit with the highest count of nests was applied (this resulted in 24 counts from 2022 and 6 counts from 2023 being removed). In some records, surveyors provided multiple abundance counts for a single survey of a

single tetrad. If it could be confirmed (using surveyors' notes or in discussion with survey organisers) that these counts referred to multiple rookeries within the tetrad, we used the sum of counts as the tetrad abundance. Otherwise, we used the highest number as the tetrad abundance. Two records were sufficiently unclear on this point that we excluded them from further analysis. Some tetrads selected for surveying were not fully surveyed due to their being partially marine or because parts of the tetrad were inaccessible. Survey coverage was categorised using the same system as in the 1996 survey (Marchant and Gregory 1999):

- 1 = complete report, full coverage of tetrad;
- 2 = complete report, from partial coverage or local knowledge;
- 3 = incomplete report from partial coverage;
- 4 = no report (no coverage);
- 5 = coverage not requested as part of sample.

For both Rook surveys (1996 and 2022/23), analysis was restricted to data from category 1 (records were inferred to be 1 unless otherwise stated). For the 2022/23 survey dataset, 16 tetrads in category 2 were excluded from further analysis. The remaining 676 (98%) of surveyed tetrads were deemed suitable for analysis.

Welsh tetrad data (including tetrads on the Wales-England border (Table 2); from the 1996 survey (Marchant and Gregory 1999) were similarly categorised, with 30 tetrads classified as category 2 or higher and excluded from analyses, leaving 174 records or 85% of the Welsh sample to be included in this analysis.

Rookeries that spanned multiple tetrads were allocated to the tetrad with the most nests in it. It should be noted that, of the 1996 data included in the abundance model of the Random Forest modelling approach, only 40 tetrads had non-zero abundance (out of 174 surveyed tetrads retained; Table 2; model detailed further below). This was considered sufficient to use this modelling approach, but contributed to the decision to rarefy the 2022/23 survey dataset for the purpose of direct comparisons between surveys (see Section 2.1, Random Forest modelling approach, below).

For abundance modelling, the total abundance represented the sum of nests per tetrad (across all rookeries, if more than one was present in a single tetrad). By contrast, the analysis of rookery size used the abundance of nests in individual rookeries.

Environmental covariates

In order to predict the occurrence of breeding Rook in tetrads and to estimate Rook abundance in Wales, a variety of different environmental covariates were summarised at tetrad level for all tetrads in Wales (including Welsh-English border tetrads when the largest proportion of their area fell in Wales; Table 2). These were:

NRW Area Statement areas (https://datamap.gov.wales/layers/inspire-nrw:NRW_AREAS). Where tetrads crossed spatial boundaries, they were assigned to the Area Statement area in which the largest proportion of their area fell.

WOS Recording Areas (<https://birds.wales/counties-map/>). Where tetrads crossed spatial boundaries, they were assigned to the Recording Area in which the largest proportion of their area fell.

CEH Land Cover Maps. These included Land Cover Map 2000 with data provided at 25m resolution used in the analysis of 1996 survey dataset (Fuller *et al.* 2002) and Land Cover Map 2021 with data

provided at 10m resolution used in the analysis of 2022/23 survey dataset (Marston *et al.* 2022). Occurrence of each habitat was calculated as a percentage of land cover per tetrad. Semi-natural and improved grassland land cover classes were also combined into a single ‘permanent grassland’ class, as well as retained in their own right.

Mean land elevation (EEA 2016; 30m resolution). The mean of the individual elevation values from all relevant land areas (i.e. excluding pixels at sea) was calculated for each tetrad.

NRW National Inventory of Woodlands and Trees (NIWT; https://datamap.gov.wales/layers/inspire-nrw:NRW_NIWT), classes: Broadleaved, Coniferous, Mixed, Young Trees. The combined coverage of these individual categories was also calculated.

Botanical Society of Britain and Ireland (BSBI) presence-absence data (Stroh *et al.* 2023) covering two time periods (1987-1999 and 2000-2019) for five key tree species for rookeries based on Rook historic use of Elm and a recent Welsh survey (Haycock and Haycock 2024). These species were Sycamore *Acer pseudoplatanus*; Beech *Fagus sylvatica*; Ash *Fraxinus excelsior*; Wych Elm *Ulmus glabra*; English Elm *Ulmus procera*. Data were converted to binomial (zero or one) data based on detection in each tetrad during surveys for Plant Atlases.

Climate data from the HadUK gridded dataset (Met Office *et al.* 2023), including annual rainfall (mm), annual temperature at surface (°C), and wind speed at 10m above ground level (m s^{-1}). Average annual rainfall, temperature and wind speed were calculated for each tetrad from the 10 years preceding each survey (i.e. 1987-1996 and 2013-2022), except in the case of wind speed for the most recent survey which was based on data from 2013-2021 (data for 2022 not available at time of analysis).

Calculation of population estimates (Random Forest model analysis)

Each survey dataset (1996 and 2022/23) was analysed independently in R version 4.2.2 (R Core Team 2022) using the ‘randomForest’ package (Liaw and Wiener 2002). Since Rooks are colonial nesters, a hurdle modelling approach was implemented, first modelling the occurrence of Rook nesting in each tetrad as a binomial (i.e. presence/absence) variable, and subsequently modelling the abundance of Rook nests within the subset of tetrads where they occurred.

The covariates included in each hurdle model were selected using a backwards stepwise elimination approach using the package ‘VSURF’ (Genuer *et al.* 2015), which is the most appropriate method for models with large numbers of independent variables (Speiser *et al.* 2019). This method assumes there are variables in the dataset that are unnecessary and ordinarily undergoes a three-step method to select the final variables, where:

Step 1 – eliminates variables the analytical process deemed irrelevant/nuisance variables from all environmental variables included (see Appendix 3 Table 1 for variables excluded).

Step 2 – selects the variables most relevant to the dependent variable.

Step 3 – refines further to produce a minimum variable list for prediction with acceptable accuracy (Genuer *et al.* 2015).

In this analysis, we were most interested in maximising the predictive capacity of the models. This is effectively achieved through step 1 of the process, since steps 2 and 3 winnow the explanatory variables to a minimum set that retains acceptable (but not maximal) predictive performance. We therefore disregarded steps 2 and 3, and retained all relevant variables in the final models. These variables are

included in the first and second hurdle models for 2022/23 and are set out in Appendix 3 Table 1.

The resulting models predicted Rook distribution (the probability of at least one Rook nest occurring in a tetrad) and abundance (the expected number of Rook nests in an occupied tetrad) for all tetrads in Wales. These results were combined by multiplying the probability of a tetrad being occupied (model 1) by the predicted abundance (model 2) to produce a predicted Rook abundance per tetrad. The probability of occupancy and overall predicted abundance of Rook nests were used to create summary maps at tetrad resolution for Wales, and regional maps for each NRW Area Statement area and WOS Recording Area.

The median and 95% confidence intervals of both the predicted number of occupied tetrads and the abundance of Rook nests were individually calculated at each output scale (whole of Wales, each NRW Area Statement area and each WOS Recording Area) using a bootstrapping approach (Efron and Tibshirani 1986). For each bootstrap run, a single tetrad (occupancy prediction or predicted count for the tetrad) was selected at random (with replacement) from the 2000 fitted trees, and the predicted values for all tetrads summed at each output scale. This process was repeated 1,000 times to yield 1,000 population estimates per output scale. From these 1,000 estimates, the median and 95% confidence intervals (i.e. the 25th, 500th and 975th values after ordering) are presented for each output scale.

Model performance testing

Two forms of model performance testing were carried out. To assess model performance when predicting occurrence and abundance of Rook at a tetrad level from the training dataset, R-squared and Root Mean Squared Error (RMSE) were calculated for all three datasets (2022/23 whole dataset, 1996 and 2022/23 rarefied dataset, note, rarefied dataset defined in the section below). The 2022/23 model output was tested against an independent dataset to assess performance. For this purpose, we used data collected for the Pembrokeshire Rook census (Haycock and Haycock 2024), which took place concurrently to, and using the same methods as, the WOS survey. This dataset surveyed all rookeries in Pembrokeshire, so Rook occurrence was expected to be equivalent to the random sample that comprised our training dataset. Since colonial nest distribution in Rook could potentially lead to uneven distribution in the landscape that is challenging to predict at tetrad-scale, we tested our model’s performance in spatial clusters of 20-25 adjacent tetrads (as opposed to individual tetrads), using a sliding envelope approach to select tetrads in a 5 x 5 grid centred on each tetrad in the testing dataset in turn. Clusters of 25 were selected because just under 4% of tetrads in the testing dataset contained counts of >100 Rook nests, so on average each cluster should contain one rookery of this large size. Where clusters had fewer than 20 tetrads (i.e. where the central tetrad was on or near the coast), the clusters were excluded from testing; however, the vast majority of tetrads were included in at least one retained cluster: only 20 out of 490 tetrads were not included in any retained clusters, all of which were offshore islands. Both observed and predicted counts of Rook nests were respectively summed within each cluster. The relationship between observed and predicted abundances at cluster-scale was visually inspected and tested for fit to the desired 1:1 relationship using a Model II regression (with Major Axis approach).

We could not repeat this test for the 1996 model outputs since no suitable independent dataset was available. However, the success of the 2022/23 model test clearly demonstrated the suitability of this approach to accurately estimating Rook populations at regional scales.

2.1. Comparisons between 1996 and 2022/23

Random Forest modelling approach

Survey effort (coverage) in the 1996 survey was lower than the 2022/23 survey (Table 1 p.38). Furthermore, the 1996 survey appears, by chance, not to have included any tetrads containing large

Output Scale		Number of tetrads surveyed (used in this study's analysis)		Number of tetrads available (including those on the Welsh-English border)
		1996 survey	2022/23 survey	
Whole of Wales (National)	Whole of Wales (National)	174	676	5,618
NRW Area Statement areas	Mid Wales	59	213	1,782
	South West Wales	39	128	1,353
	North West Wales	38	153	1,255
	North East Wales	15	82	469
	South East Wales	11	54	414
	South Central Wales	12	46	345
WOS Recording Areas	Pembroke	13	56	490
	Denbigh	20	85	472
	Montgomery	17	62	520
	Ceredigion	13	50	476
	Carmarthen	21	47	622
	Gwent	11	52	380
	Caernarfon	12	54	435
	East Glamorgan	12	48	379
	Anglesey	7	39	232
	Flint	4	33	160
	Radnor	11	30	307
	Brecon	16	68	449
	Meirionnydd	12	27	456
	Gower	5	25	240

Table 1. Number of tetrads used in this study’s analysis (after data processing) surveyed at each output scale for the 1996 and 2022/23 surveys. Also presented is the total number of tetrads available per output scale, including some Wales-England border tetrads. Note, the one additional opportunistically-surveyed tetrad is not included in this table but is found in the South West Wales NRW Area Statement area and in the Pembrokeshire WOS Recording Area.

rookeries, and rather few medium sized rookeries. As a consequence of this apparent skew, the range of observed values was substantially smaller than that of the 2022-23 survey, which means any population estimate calculated from the 1996 survey will likely be an underestimate driven by tetrad selection. The Random Forest modelling approach is unable to predict beyond the range of observed data. In order to limit the effect of this skew in the historic survey dataset affecting our estimate of abundance change since 1996, we similarly skewed the 2022/23 survey data using a rarefaction approach. The selection provided a similar number of data points to 1996, with a similar range of observed values, by constraining the rarefaction to exclude tetrads surveyed in 2022/23 with higher Rook abundance than the maximum abundance (112 nests) recorded in the 1996 survey. The resulting rarefied 2022/23 dataset had a similar frequency distribution of abundances to the 1996 dataset (Appendix 4 Figure 1), in principle allowing us to re-run the same Random Forest modelling approach as described above (covariables included: Appendix 3 Table 1), in order to make direct comparisons between the two surveys. Nevertheless, given these concerns, we place greater confidence in Rook abundance change estimates produced by the BBS than those estimated by comparison between these two models.

Statistical comparisons

Differences in tetrad occupancy and Rook nest abundance between the 1996 and 2022/23 surveys were tested by fitting generalized linear mixed-effects models (GLMMs) using the package glmmTMB (Brooks *et al.* 2017), with the significance of these differences evaluated using Likelihood Ratio Tests. To assess change in distribution, we modelled the effect of survey (period) on the predicted probability of Rook occurrence for each tetrad derived from the 1996 model and full 2022/23 model (rather than the rarefied 2022/23 model), with tetrad included as a random effect, and assuming a binomial error distribution. To assess change in abundance, the predicted abundance of Rook nests for each tetrad (using the rarefied 2022/23 model) was the dependent variable, and survey (period) the independent variable, with tetrad included as a random effect, using a zero-inflated negative binomial error distribution (with a log link function).

2.2. Rookery size

Rookery sizes were compared between 1996 and 2022/23 surveys by fitting and testing a generalised linear model (GLM). The abundance of breeding pairs (per rookery, rather than per tetrad, and therefore by definition excluding counts of zero) was fitted as the dependent variable with survey period as the independent variable, using a negative binomial error distribution (with a log link function). Whilst a GLMM approach that directly compared the same rookeries counted in both surveys might have had greater statistical power, only eight tetrads with Rook present were surveyed in both periods, which was insufficient for such an analysis.

3. Results

3.1. Survey coverage

The final datasets after processing consisted of 676 tetrads, of which 199 included at least one Rook nest from the 2022/23 survey (29% of surveyed tetrads were occupied); and 174 tetrads, of which 40 included at least one Rook nest from the 1996 survey (22% of surveyed tetrads were occupied). Full details of numbers of tetrads surveyed for each output scale are detailed in Table 1.

3.2. Population estimates

The Random Forest 2022/23 whole dataset combined model explained 75.35% of the variation in Rook breeding pair counts at a tetrad scale (RMSE = 11.90; Appendix 5 Table 1). Model testing comparing total abundance for 25-tetrad clusters between the independent dataset (Haycock and Haycock 2024) and predicted Rook abundance from our 2022/23 whole dataset model suggested the two were very closely aligned (Slope: 0.992, 95 % CIs: 0.904-1.090; Intercept: 76.894, 95 % CIs: 42.431-108.302; R2: 57.00 %; P-perm < 0.01; Appendix 5 Figure 1).

The Welsh Rook breeding population was estimated to be **44,127 pairs** (95% CI: 41,144-46,940). At smaller spatial scales, Mid Wales Area Statement area had the greatest abundance at 13,064 breeding pairs (95% CI: 11,651-14,563). Pembrokeshire had the greatest abundance of any WOS Recording Area (estimated abundance 8,238 breeding pairs, 95% CI: 6,747-9,808) and highest density per km² (5.1 breeding pairs per km²). Combined, the mid Wales and South West Wales Area Statement areas hold 58.4% of the Welsh Rook breeding population. There were relatively fewer Rooks in central Wales compared with either coastal regions or the English border, with particular hotspots in abundance in Pembrokeshire (South West Area Statement area), Montgomeryshire (eastern Mid Wales Area Statement area), and Denbighshire/Flintshire (North-east Area Statement area; Flintshire = 3.8 breeding pairs per km²). Full results are presented in Table 2 and Appendix 8 Table 1 and Figures 1a-b (p.41) and 2a-b (p.42), and for regional maps see Appendix 6.

3.3. Correlations between Rook breeding pair abundance and environmental variables in 2022/23
The variables with greatest influence on 2022/23 Rook breeding abundance (determined by VSURF Step 2) were latitude (easting), rainfall, and habitat classes including built-up areas, arable, improved

Output Scale	Output Scale	Median	Lower CI	Upper CI	Output Scale area (km2)	Density (number breeding pairs per km²)
Whole of Wales (National)	Whole of Wales (National)	44,127	41,144	46,940	21,225	2.1
	Mid Wales	13,064	11,651	14,563	7,001	1.9
NRW Statement areas	South West Wales	12,708	10,951	14,527	4,962	2.6
	North West Wales	6,968	5,926	8,151	4,525	1.5
	North East Wales	5,918	4,930	7,003	1,840	3.2
	South East Wales	3,034	2,337	3,846	1,616	1.9
	South Central Wales	2,261	1,645	3,035	1,281	1.8
	Pembroke	8,238	6,747	9,808	1,625	5.1
WOS Recording Areas	Denbigh	4,591	3,777	5,613	1,853	2.5
	Montgomery	4,575	3,759	5,463	2,062	2.2
	Ceredigion	4,105	3,310	4,990	1,806	2.3
	Carmarthen	3,148	2,480	3,914	2,444	1.3
	Gwent	2,875	2,207	3,677	1,473	2.0
	Caernarfon	2,522	1,922	3,295	1,505	1.7
	East Glamorgan	2,427	1,768	3,207	1,412	1.7
	Anglesey	2,270	1,723	2,953	746	3.0
	Flint	2,198	1,586	2,917	579	3.8
	Radnor	2,139	1,609	2,754	1,220	1.8
	Brecon	2,049	1,481	2,693	1,793	1.1
	Meirionnydd	1,378	919	1,943	1,772	0.8
	Gower	1,327	857	1,918	871	1.5

Table 2. Predicted abundance of Rook breeding pairs at different scales, based on models of the 2022/23 survey data, median and lower and upper 95 % confidence intervals (lower = 2.5 % and upper = 97.5 %). The area (km²) of each output scale is also presented alongside a calculated density of Rook breeding pairs per output scale area (median count/area).

grassland, permanent grassland, broadleaf woodland (LCM); and overall woodland cover (NRW NIWT). The full list of variables are presented in Appendix 3 Table 2; and figures showing relationships between covariates and predicted abundance in Appendix 7. Other variables retained for prediction but with less impact on model fit were, longitude (northing), elevation, temperature, wind speed; coniferous woodland, mountain/heather/bog, freshwater (LCM); broadleaf woodland, coniferous and young trees (NRW NIWT); and English Elm distribution (Appendix 3 Table 1).

Predicted Rook breeding density (number of Rook breeding pairs per tetrad) was negatively related to mean elevation, rainfall and wind speed, but positively related to mean temperature (Appendix 7 Figure 1). Predicted abundance was positively associated with the area of arable land, improved grassland and permanent grassland, but negatively associated with the area of all other land cover types, including woodland (Appendix 7 Figures 2 and 3). Predicted abundance was higher in tetrads where the five tree species associated with nesting Rooks had been recorded, with this association being strongest for Wych Elm and English Elm (Appendix 7 Figure 4).

3.4 Changes in distribution and abundance between 1996 and 2022/23

Rook distribution (the number of occupied tetrads in Wales) has declined by 5.6%, although the probability of any given tetrad being occupied has not significantly changed, P = 0. 0932; Table 3; Figure

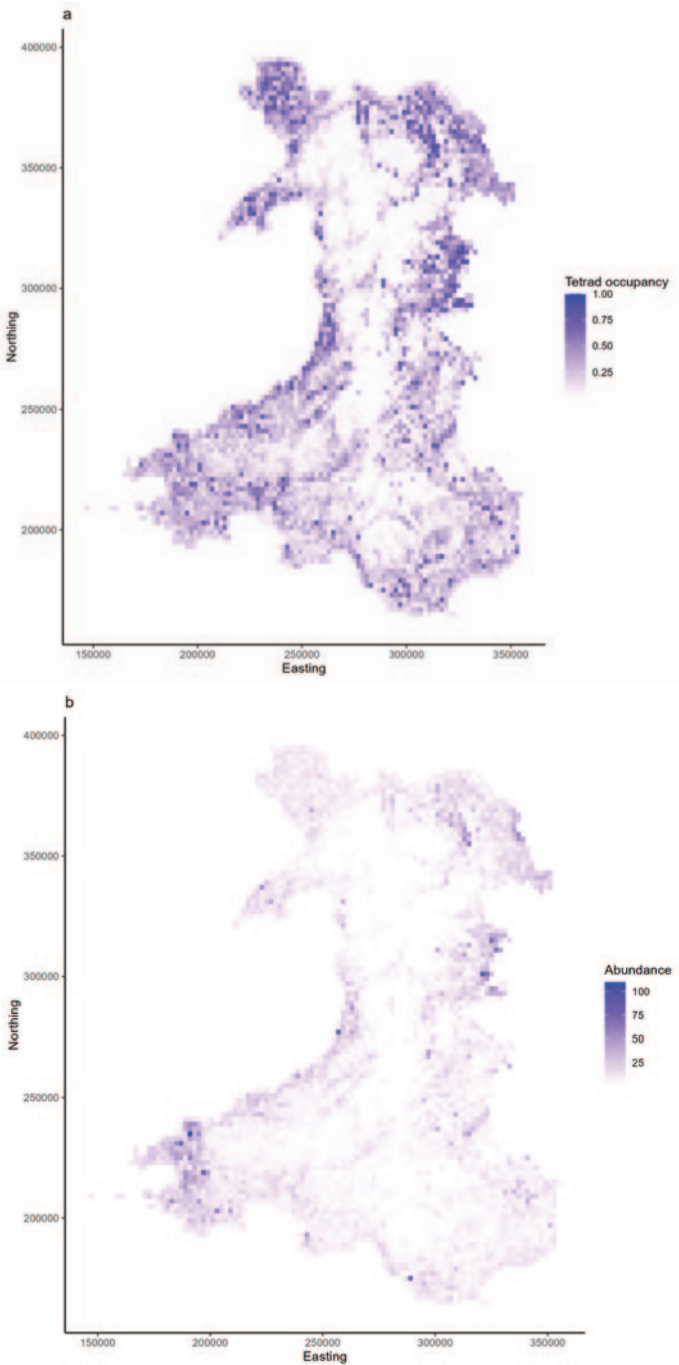


Figure 1. Modelled predictions for

a) the probability of tetrad occupancy by Rook (i.e. the probability of a Rookery being present in each tetrad), and

b) abundance of Rook breeding pairs per tetrad. Note: darker blue means a higher probability of occupancy and a higher abundance of nests respectively.

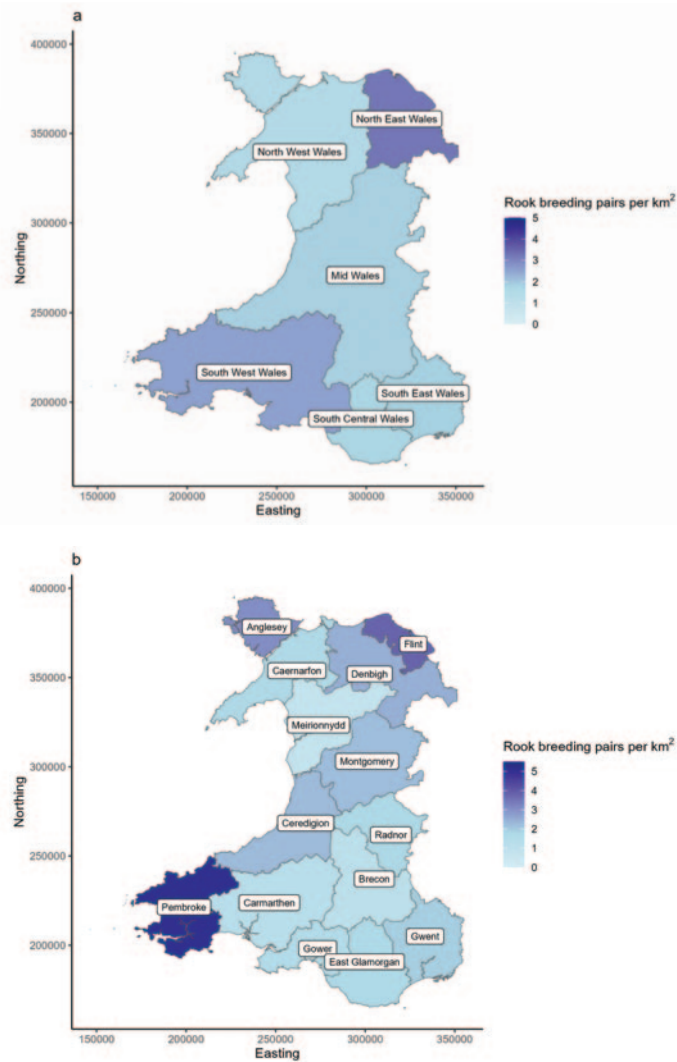


Figure 2. Mapped Rook breeding pair density per km² where darker colours equate to a higher density, for
a) the six terrestrial NRW Area Statement areas and

b) the 14 WOS Recording Areas. See Table 2 for full details of WOS Recording Area names.

3a. Abundance per tetrad has declined significantly ($P < 0.0001$; Table 3; Figure 4a), suggesting a national population decline of 19% since the 1996 survey, which is likely to be an underestimate (see Discussion).

There was substantial regional variation in both distribution and abundance change. Distribution increased in four of the six terrestrial Area Statement areas and seven out of 14 WOS Recording Areas, but these increases were more than offset by larger decreases in South West and South East Wales. Similarly, a marked increase on Anglesey contrasts with large decreases in Pembrokeshire, Carmarthenshire and Gwent (Figure 3a-b, p.43).

	Error distribution of the model	Effect size	Standard error	LRT	P
Presence/ absence	Binomial	-0.075	0.045	2.820	0.0932
Abundance	Zero inflated negative binomial GLMM, Link function = log	-0.184	0.011	272.75	< 0.0001
Rookery size	Negative binomial, link function = log	-0.222	0.127	3.213	0.0731

Table 3. Descriptions of models comparing a) occupied tetrads between 1996 and 2022/23, b) abundance of rook nests between 1996 and 2022/23, and c) rookery sizes.

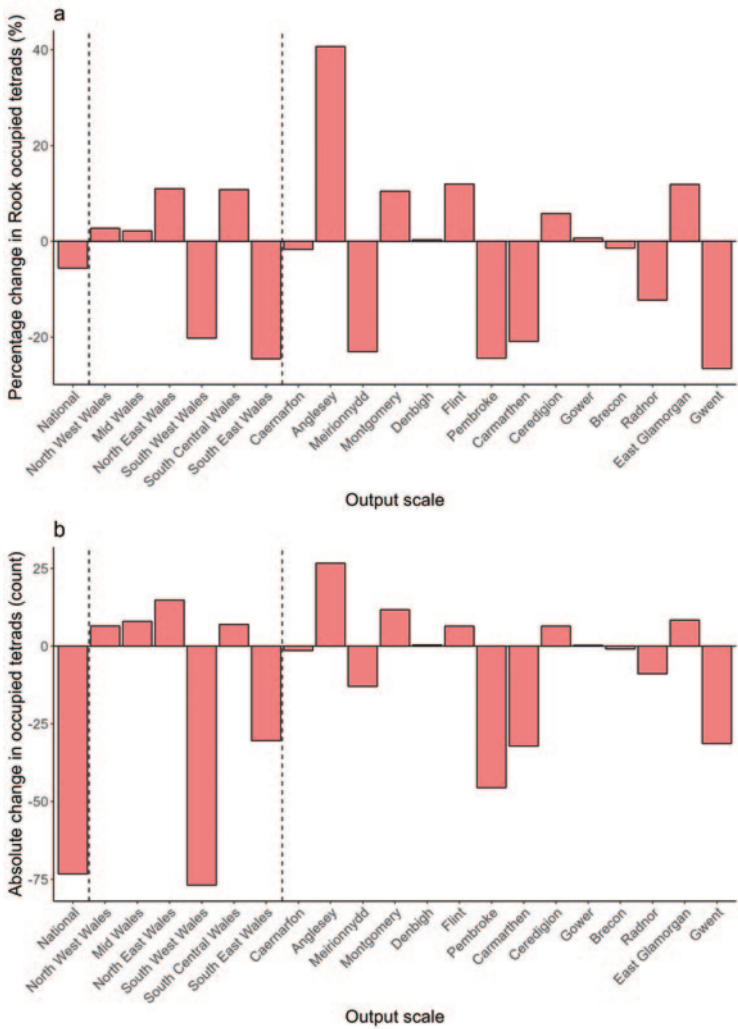


Figure 3. Change in tetrads occupied by Rook nests, from 1996 to 2022/23 (whole dataset) expressed as a) percentage change and b) absolute change in number of occupied tetrads. See Table 2 for full details of WOS Recording Area names.

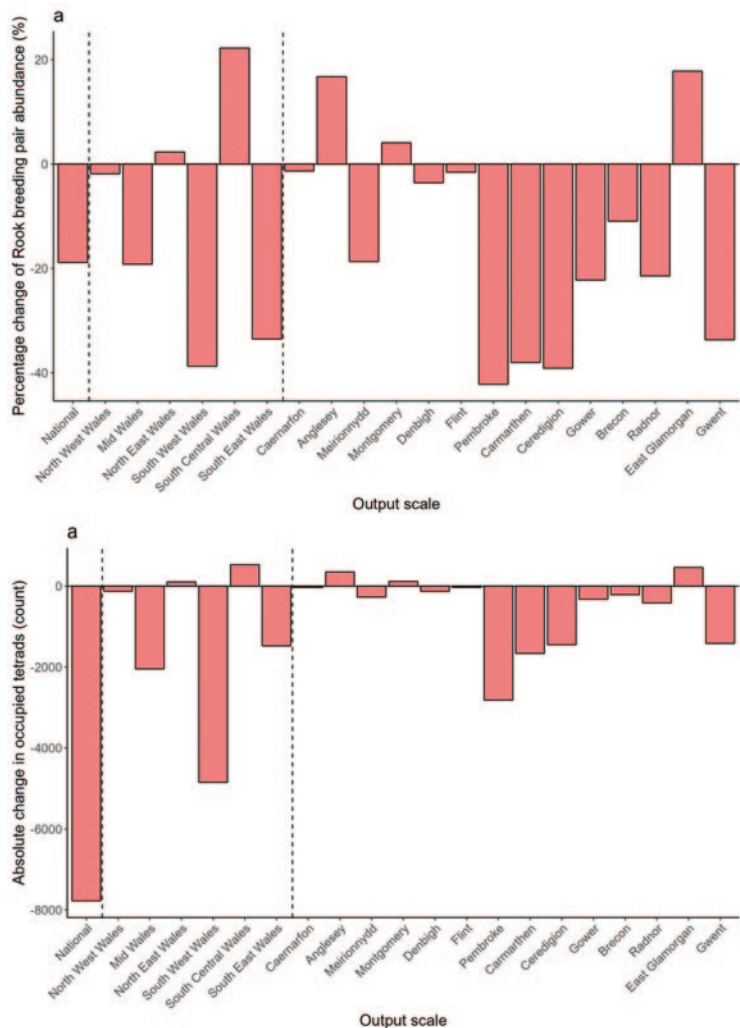


Figure 4. Rook breeding pair abundance change, from 1996 to 2022/23 (rarefied dataset), expressed as a) percentage change and b) absolute change in number of Rook nests. These figures show effect sizes as estimated in this study; note that the Welsh Rook national estimate of population change is considered to be an underestimate (19 % decline, compared to 63 % decline calculated from BBS indices), and change at regional scales is also likely to be impacted, with declines underestimated and increases overestimated. See Table 2 for full details of WOS Recording Area names.

Rook nest abundance declined in the majority of areas, including four of the six Area Statement areas and 11 of the 14 WOS Recording Areas. Particularly marked declines were seen in South West and South East Area Statement areas and Pembrokeshire, Carmarthenshire, Cardiganshire (Ceredigion) and Gwent WOS Recording Areas. In the few Area Statement areas and WOS Recording areas where Rook

breeding pair abundance did increase, the scale of those increases was very small in comparison to the losses elsewhere (Figure 4a-b, p.44).

3.5. Change in rookery size

The average number of breeding pairs in rookeries declined between 1996 and 2022/23, from 25 breeding pairs in 1996 (CI: 22–28) to 20 breeding pairs in 2022/23 (CI: 19–21), a decline of 20% (Appendix 3 Table1). This change was not statistically significant when tested ($P = 0.0731$).

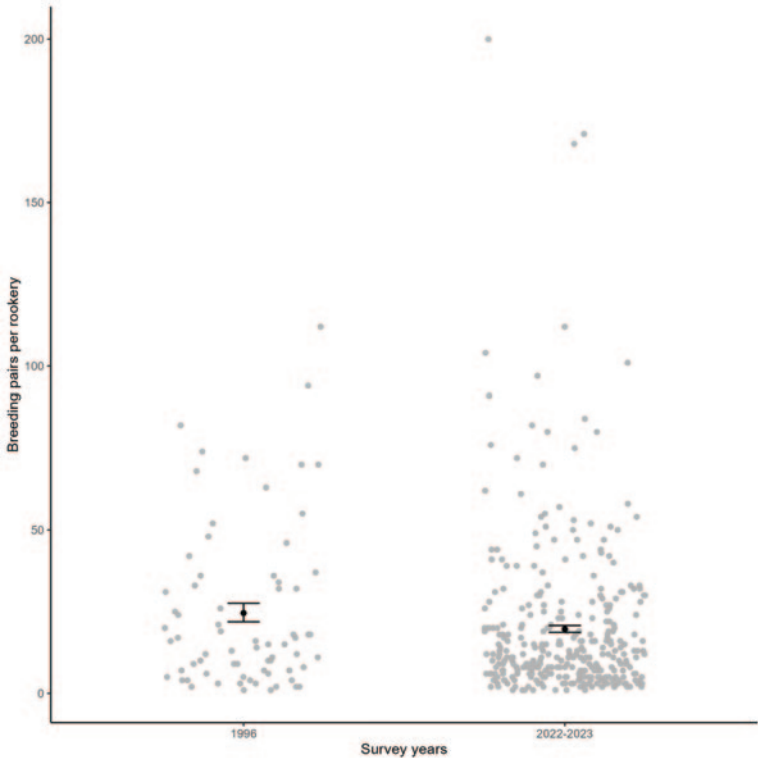


Figure 5. Change in number of breeding pairs in rookeries between 1996 and 2022/23. Model estimates for the average number of breeding pairs in rookeries are shown by black circles (error bars show 95 % CI) and all observed rookery number of breeding pairs in rookeries are shown by grey circles.

4. Discussion

Overall population change

We estimate that the Welsh breeding population of Rook in 2022/23 was **44,127 pairs** (95% CI: 41,144–46,940). This represents a decline in both abundance and range (tetrad occupancy -5.6%) since 1996. This overall decline is the sum of substantial regional variation in both distribution and abundance change at different spatial scales. The average size of rookeries has also declined by 20% since 1996.

Rook have declined most in the South West Area Statement area and Pembrokeshire WOS Recording areas, both proportionally and in absolute terms. At the same time, some areas have seen increases in breeding numbers of Rook; these include South Central Wales (Area Statement area) and Anglesey

(WOS Recording Area).

Despite the national decline, there are still some stronghold areas with significant populations of Rooks, most notably in Pembrokeshire (South West Area Statement area, despite the aforementioned decline); Montgomeryshire (eastern Mid Wales Area Statement area), and Denbighshire/Flintshire (North East Area Statement area).

A useful cross-check of the modelled outputs presented in this report is provided by the full-coverage survey delivered by Pembrokeshire Bird Group, independently and in parallel with the nationally coordinated stratified randomly selected tetrads analysed here. This additional dataset represents a full census of rookeries in the Pembrokeshire WOS Recording Area (Haycock and Haycock 2024). That census approach estimated Pembrokeshire contained a minimum of 6,679 Rook nests, with an estimated likely total of 6,780–6,800 nests. This overlaps with the confidence interval for our modelled estimate for Pembrokeshire, of 8,238 (6,747-9,808) nests.

Abundance and distribution change

This analysis estimated a 5.6% decline in tetrad occupancy since 1996, which is comparable to the change recorded between 1990–2010 by successive Bird Atlases (Balmer *et al.* 2013). Modelled BirdAtlas outputs are generally reported using a larger 10 x 10 km (hectad) scale, and using this spatial resolution the Rook distribution has declined by four hectads, compared to a net decrease of one hectad between the breeding Atlases (Appendix 9 Figure 1). The spatial patterns of change between these two independent but temporally overlapping pairs of surveys were broadly similar; both highlighting range losses in the central band of Wales contrasting with expansions along the coast and English border.

Comparison between the abundance change analysis and the BBS trend

The models fitted in this study suggest a 19% decline in Rook breeding abundance at national scale in Wales since 1996. Whilst this is a substantial decline, it is less severe than the 59% decline calculated from BBS trend data (difference calculated between the smoothed BBS indices for 1996 and 2022; Heywood *et al.* 2024). This difference is likely to be the result of small sample size, and apparent skew, in the 1996 survey data. The 1996 Rook survey sampled only 40 tetrads containing rookeries in Wales, with a maximum rookery size of 112, while the largest rookery counted in the 2022/23 survey contained 200 nests. Because the Random Forest approach is not capable of predicting beyond the observed range of values, this prevented a robust estimation of population size for Wales in 1996 (that survey was designed to deliver a UK population estimate rather than a regional one). We attempted to compensate for this effect using sample rarefaction, which partially resolved the issue – sufficiently at least to allow for testing whether population change was significant, and so that relative differences in population change could be estimated at regional scales. We derive two clear messages from this analysis, which are worthy of note.

First, despite the potential for colonial species' distributions to be more aggregated than is ideal for the BBS monitoring approach, that survey remains the best indicator of overall population change for Rook in Wales; especially since BBS records individual birds rather than nests, and therefore includes non-breeding birds where they are present. In a declining population, reduced non-breeding birds (sub-adults or adult birds failing to breed in the survey year) may reflect the processes of population change, for example lower productivity reducing sub-adult numbers, or habitat quality alterations reducing the survival of adult birds. Second, caution is indicated in comparing between historic and contemporary surveys with very different sample sizes, even where survey methodology is consistent. Analytical compensation for sample size cannot compensate for the risk of skew or bias generated by the smaller sample size. However, it is relevant to note here that these analytical, ecological or sample-related issues are relevant only to the comparison of population size between 1996 and 2022-23; the analyses of change in distribution and average rookery size are not affected by these issues and the estimated

5.6% decline in range and 20% decline in rookery size are considered analytically robust.

The breeding Rook population decline in Wales includes considerable regional variation which we were able to estimate using the rarefaction approach, with particularly marked declines in abundance in the South West Area Statement area and Pembrokeshire WOS Recording Area and increases in South Central Wales Area Statement area and Anglesey WOS Recording Area. However, the described limitations of the 1996 dataset and Random Forest approach mean that we are likely to have underestimated declines and overestimated increases in abundance at regional scales. Nevertheless, our modelled declines do reflect those observed in other field studies, in particular the Pembrokeshire WOS Recording Area complete census, which extended this study's survey to cover every rookery in Pembrokeshire, as described above (Haycock and Haycock 2024).

Possible drivers of change

Changes in farming practices and pesticide usage, along with persecution, have previously been identified as drivers of Rook population decline (Brenchley 1986; Charter 1996; Kasprzykowski 2007; Poluda *et al.* 2023; Brezghunova and Sinna 2024). Furthermore, Dutch Elm disease and the consequent decline in preferred nesting trees had impacts on the abundance and distribution of nests and rookeries (Osborne 1982; Sage and Whittington 1985). In addition, other studies have noted that nesting Rook are vulnerable to predation of eggs or chicks, especially where the predatory species are novel – including invasive or colonising species, re-introduced or recovering species and those experiencing range shifts. In Wales, recent recoveries in Raven *Corvus corax* and Red Kite *Milvus milvus* might be considered novel predatory pressures on breeding Rook, amongst others (Slater 2023).

Another likely pressure on Rook populations is contemporary changes in farming practice, some of which may be positive. For example, increases in livestock densities on grassland may increase some invertebrate populations and decrease sward height, promoting foraging and increasing food availability for invertebrate feeders (Charter 1996, Devereux *et al.* 2004). Alongside land and crop management, the measurable impacts of climate change on the environment are increasing and will likely become more important for Rooks and other species (IPCC 2023).

These possible drivers of change are clearly reflected in the retained environmental variables from this analysis. Extreme weather impacts (especially changes in precipitation and heatwaves; IPCC 2023) will become increasingly important to the species since their colony locations are sensitive to elevation, rainfall, temperature and wind speeds. The Rook colony climate envelope, in common with other breeding species, may move progressively higher as climate change advances (Tingley *et al.* 2012), whilst the tree species and tree maturity Rook select for colonies may not be able to keep pace with the change. Avoidance of higher elevations combined with a lack of suitably large trees for nesting on higher ground may mean Rooks are unable to track their climate envelope as it shifts uphill.

The most important habitats associated with Rook abundance were arable habitat followed by grassland habitats (at tetrad scale). Continued availability of arable and grassland foraging habitats (Collinge 1921; Holyoak 1972; Feare 1978; Charter 1996) is likely to remain important for Rook, as is their avoidance of woodland. Although the pace of change is slow, national targets for woodland planting, commercial conifer forestry and rewilding agendas may also conflict with Rook foraging habitat in future.

Finally, a small number of tree species continue to be important for supporting rookeries, especially Wych Elm *Ulmus glabra* and English Elm *Ulmus minor*. This ecological preference persists despite the catastrophic declines in both tree species associated with a novel tree disease, Dutch Elm Disease (*Ophiostoma novo-ulmi*; Forest Research 2024) from Canada in the late 1960s. Subsequent studies have demonstrated that Rook can successfully switch to other tree species if necessary, but continue to nest in suitably large Elms if these are present (Osborne 1982; Sage and Whittington 1985). It is notable that

despite its prevalence in Wales, Oak (*Quercus sp.*) could be excluded from the short-list of important tree species because there is no evidence that Rooks select or prefer this species in Wales (Osborne 1982), although its near-ubiquitous distribution in Wales would also have made the species a poor spatial predictor in models of Rook population density.

A final possible historic driver of population change in Rook is their inclusion on Schedule II part II of the Wildlife and Countryside Act from 1981, with inclusion on the General Licence (GL001) prior to 2024, permitting them to be lethally controlled for the prevention of damage to crops, livestock or food. Numbers or rates of lethal control under the General Licence were not required to be monitored or reported in any way, and it is impossible to know how many Rooks were removed in this way. Their removal from General Licence 001 (Licence to kill or take certain wild birds to prevent serious damage to livestock, foodstuffs for livestock, crops, vegetables or fruit or to prevent the spread of disease to livestock, foodstuffs for livestock, crops, vegetables or fruit) in 2024 was largely a result of their declining conservation status..

Conclusions

We estimate the Welsh breeding population of Rook in 2022/23 to be **44,127 pairs** (95% CI: 41,144-46,940). In addition, we provide new spatial distribution and abundance maps for Wales, and for two smaller spatial scales commonly used in biodiversity reporting (each of the six terrestrial NRW Area Statement areas and the county-level WOS Recording Areas). These resources may be key to understanding future change, as they provide a robust population baseline for future survey comparisons.

This population estimate reinforces other monitoring data in highlighting Rook population and range decline of -5.6% (range) and -20% (mean rookery size) since 1996. For reasons associated with the small survey sample in Wales in 1996, the best indicator of population change remains the Breeding Bird Survey (BBS) index, which suggests a population decline of -59% over a similar period.

Rook nest abundance and rookery size have both declined since the 1996 survey. These changes reflect wide pressures on the Rook breeding population which are poorly understood, not controlled and likely cumulative and very broad-scale. The conservation implications of these pressures on the species in Wales are clear, and highlight the importance to the species of anthropogenic land management and farming practices, licensed control and the accelerating progress of climate change in Wales.

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Ethical Statement

There were no ethical concerns associated with this study.

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All appendices may be found online at:
<https://birds.in.wales/wp-content/uploads/2024/10/Hereward-et-al.-RookReport.-Milvus.-NEW-Appendicesv2-1.pdf>

The Ely valley: an important site for wintering Green Sandpiper *Tringa ochropus*

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Key words: Population; Green Sandpiper; Wales; protected site; threats.

Abstract
Green Sandpipers winter in small numbers in the UK. Establishing a robust wintering population estimate for the species is difficult because they occupy a variety of habitats and are thinly spread across the country. Here we describe a survey method for Green Sandpiper, using the River Ely in Glamorgan as a case study. A series of coordinated counts undertaken by a team of local volunteers across several sections of the river during the core winter period (November to March) revealed that the River Ely regularly held at least 7-15 wintering Green Sandpipers. The survey method was designed to avoid double-counting these highly mobile birds, such that each month produced both a ‘safe’, minimum count and a possible maximum count of birds present on the river. The value of researching suitable habitat for the species is demonstrated by the presence of Green Sandpipers during the study at sites along the river where they had not previously been recorded. Variation in the monthly counts and the spatial behaviour of the birds are explored and the potential threats to species’ continued presence on the Ely are discussed. The River Ely in Glamorgan has, for many years, attracted wintering Green Sandpipers and the results of this study demonstrate that they are present there in nationally significant numbers, regularly exceeding 1% of the species’ current wintering population estimate.

Crynodeb
Mae Pibyddion Gwyrdd yn gaeafu mewn niferoedd bychain yn y DU. Mae creu amcangyfrif cadarn o’r niferoedd sy’n treulio’r gaeaf yma yn anodd oherwydd eu bod yn defnyddio amrywiaeth o gynefinoedd ac mae’r boblogaeth yn wasgaredig iawn. Yma disgrifiwn dull arolygu ar gyfer Pibyddion Gwyrdd, gan ddefnyddio’r Afon Elái ym Morgannwg fel astudiaeth achos. Datgelodd cyfres o gyfrifiadau wedi’u cydamseru gan dîm o wirfoddolwyr lleol ar draws sawl rhan o’r afon yn ystod cyfnod craidd y gaeaf (Tachwedd i Fawrth) fod yna o leiaf 7-15 o Bibyddion Gwyrdd yn gaeafu ar hyd yr Afon Elai yn rheolaidd. Cynlluniwyd yr arolwg mewn modd a fyddai’n osgoi cyfrif yr un adar, sy’n hynod symudol, ddwywaith, a bod pob cyfrif misol yn creu lleiafswm ac uchafswm posibl o’r adar oedd yn bresennol ar yr afon. Mae gwerth ymchwilio i gynefin addas ar gyfer y rhywogaeth hon yn cael ei brofi gan bresenoldeb Pibyddion Gwyrdd yn ystod yr astudiaeth ar safleoedd ar hyd yr afon lle nad oeddent wedi’u cofnodi o’r blaen. Archwilir amrywiadau yng nghyfansymiau’r cyfrifiadau misol a defnydd yr adar o wahanol rannau o’r afon a thrafordir y bygythiadau posibl i bresenoldeb parhaus y rhywogaeth hon ar yr afon hon. Mae’r Afon Elái ym Morgannwg, ers sawl blwyddyn, wedi denu Pibyddion Gwyrdd i aeafu ar hyd ei glannau ac mae canlyniadau’r astudiaeth hon yn profi eu bod yn bresennol yno mewn niferoedd o bwys cenedlaethol, gan fod mwy nag 1% o amcangyfrif o’u poblogaeth gaeafu presennol i’w gweld yno’n gyson.

Introduction
The Green Sandpiper (*Tringa ochropus*) is a medium-sized wading bird which breeds in sub-boreal areas across northern Europe as far east as Russia and the Pacific Ocean (del Hoyo *et al.* 1992-2011). The



Green Sandpiper on the River Ely: Tim Collier

majority seen in the UK breed in Fennoscandia and are observed on their autumn migration to their wintering grounds in the Mediterranean basin and in Africa (Balmer *et al.* 2013, Smith *et al.* 2024). Observations in the UK peak in July-September (BTO/RSPB/BirdWatch Ireland/SOC/WOS 2024. BirdTrack), but a small number choose to winter in Britain and Ireland at the very northwestern edge of their wintering grounds. During this period, they are usually found in ones and twos, favouring freshwater sites such as marshes, the edges of rivers, streams, lakes, reservoirs, floodplains, small pools and ditches where they can access areas of shallow water or unvegetated muddy areas to feed. They can also be found, albeit less frequently, at coastal sites such as the edges of coastal marshes and estuaries. (Snow and Perrins 1998, Pritchard *et al.* 2021).

UK and Welsh wintering population estimates

It is difficult to produce a robust wintering population estimate for this species because “it occurs thinly but widely across much of the country” (Musgrove *et al.* 2011) and many birds will remain undetected. Winter population estimates have varied greatly over the last half century, from 150-250 (Prater 1981) to an estimated 500-1,000 birds (Lack 1986). Combining the estimated totals from county bird reports and WeBS data led to an UK estimate of 910 birds in Musgrove *et al.* (2011). The most recent UK estimate is 290 birds (Frost *et al.* 2019) which used a compilation of county, regional and multiple other sources of data. This estimate is lower than Musgrove *et al.* (2011), partly due to an attempt to exclude

passage birds by restricting data to the core wintering months of November–March (previous estimates included records from October). The authors of Frost *et al.* (2019) suggest that this should be considered a minimum estimate, since many birds are likely to be overlooked. The wintering population in Wales is currently believed to be < 100–150 individuals (Pritchard *et al.* 2021).

Population change

Wintering Green Sandpiper in Wales are currently under an Amber Wetland Bird Survey (WeBS; www.bto.org/webs) alert having declined 48% in the long term (up to 25 years) and 30% in the medium term (up to 10 years). Across Britain, wintering birds have declined less steeply, at -18% in the long term, -11% in the medium term and -15% in the short term (last five years). Here, in the short term, the Welsh data diverge very notably from both long and medium-term Welsh and all UK trends, with an astonishing increase of 93% in the most recent five years, but the reason for this change is unclear (Woodward *et al.* 2019).

Site conservation status

In Britain, a wetland is considered nationally important if it regularly holds 1% or more of the estimated British population of one species or subspecies of waterbird. Using the estimate from Frost *et al.* (2019), 1% of the estimated wintering population of Green Sandpipers is three birds. In the case of small populations such as Green Sandpiper where 1% of the national population is less than 50 birds, 50 is normally used as a minimum qualifying threshold for the designation of sites of national importance (Stroud *et al.* 2021).

Green Sandpiper in the Ely Valley

The earliest species record found by the authors for the Ely valley was submitted by Col. H. Morrey-Salmon in August 1973. A small number of records were submitted through the 1980s and 90s (Glamorgan, Mid & South Glamorgan and Eastern Glamorgan Annual Bird Reports), with a maximum count during that period of 13 birds in July 1983 (Smith 1984). The number of records increased dramatically in the 21st century; probably due to increased birding effort, since the area has built a reputation as a ‘hotspot’ for Green Sandpipers with almost guaranteed sightings during passage periods and winter months. They are also occasionally recorded in the Ely valley away from the river, primarily during periods of high water, when they move onto nearby flooded areas on the floodplain to feed (pers. obs).

There are two WeBS sites along the Ely. Peterston and Pendoylan Moors (2.95km long) and River Ely at Duffryn Bach Farm (1.13km); created in March 2014 and December 2019 respectively. Both sites have been counted almost monthly throughout the year from the outset and, between them, have recorded Green Sandpipers in every month except May. Observers have recorded up to nine birds at Peterston and Pendoylan Moors during the peak passage period in July and August, and up to eight birds during the winter. At River Ely at Duffryn Bach Farm, the maximum passage count is four and maximum wintering count is five.

The Glamorgan Bird Club (GBC) holds additional data which primarily include counts from the Ely at Peterston and Pendoylan Moors, but also from other areas in the valley. Using both the WeBS and GBC datasets, Figure 1 shows the maximum annual counts of wintering Green Sandpipers (Nov to Mar) observed at single locations on the River Ely between 2001-2021. These counts demonstrate that the numbers of birds in the valley have exceeded 1% of the estimated national population threshold (3 birds) there for at least 20 years.

This knowledge prompted the authors to ponder whether birds could also be present on other, under-watched, stretches of the river; and given the declines observed in UK WeBS data, whether the area

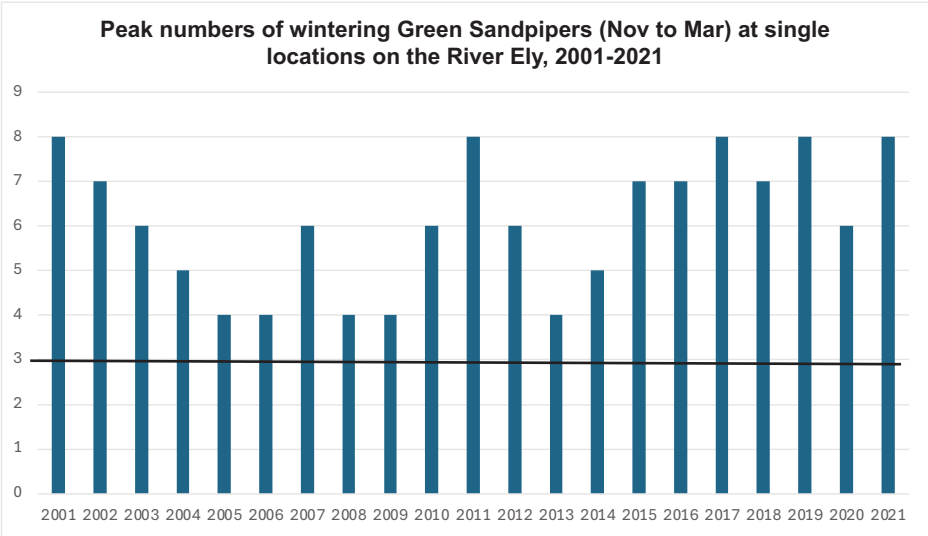
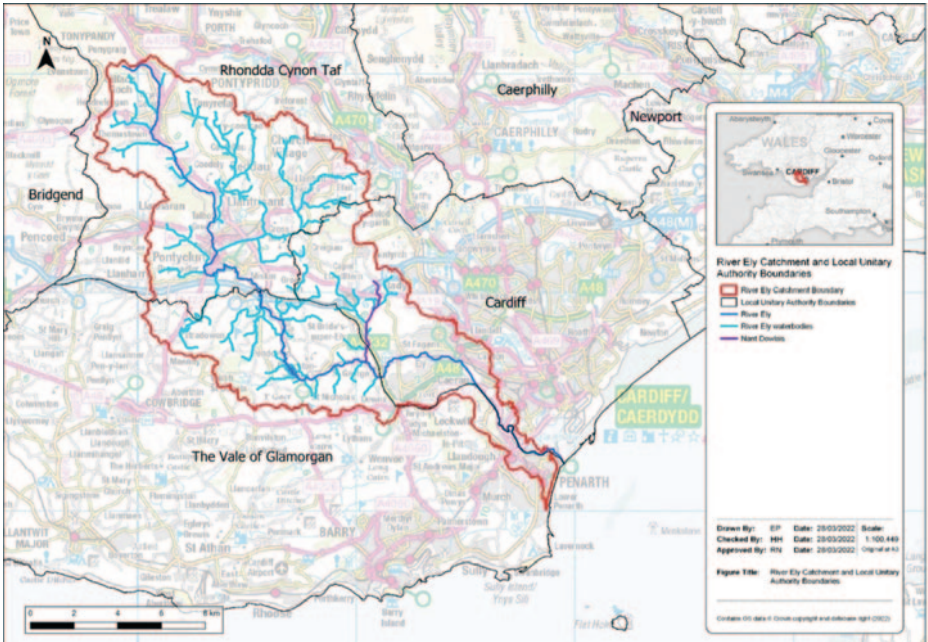


Figure 1. Maximum numbers of wintering Green Sandpipers (Nov to Mar) at single locations on the River Ely, 2001 – 2021. Includes WeBS and Glamorgan Bird Club data. Current 1% of UK wintering population estimate represented by black line.



The River Ely meanders at Peterston and Pendoylan Moors, May 2010 (© Toby Driver)



The Ely Catchment © Crown

could potentially be of national significance for the species. This study is intended to provide a more robust population estimate for the Ely using a coordinated count approach.

Methods

Study area in the River Ely catchment

The River Ely rises in the hills south of Tonypanydy, Rhondda Cynon Taf at SS9898390405. It is 39.6km (24.6 miles) long and flows into Cardiff Bay ST1852673656. The predominant agricultural activities in its catchment are dairy farming in the middle and lower reaches, and sheep farming in the upper reaches (NRA 1994). Maize (*Zea mays*) is also grown in the valley (pers obs).

Starting with the two existing WeBS sites, we used historical data, local knowledge and GoogleEarth (<https://earth.google.com/web>) to map suitable winter foraging habitat for Green Sandpiper along the river. This focused attention on 13.4km of the river’s mid-section, an area of agricultural floodplain from Miskin (ST0521080584) downstream to St Fagans (ST1193976984). One of the notable features

Sector number	Name	Boundary (upstream)	Boundary	Length
1	Miskin	ST0521080584	ST0565379686	1.48km
2	Pont-tal-y-bont	ST0565379686	ST0568279319	0.44km
3	Duffryn	ST0568279319	ST0659378048	1.13km
4	Pendoylan Moors	ST0662777390	ST0679876729	0.89km
5	Peterston Moors	ST0679876729	ST0758175992	2.3km
6	St Georges to Drope	ST1013076890	ST1087276198	0.88km
7	Drope to St Fagans	ST1116276144	ST1146376688	0.76km
8	St Fagans	ST1146376688	ST1193976984	0.52km

Table 1. The study’s sector numbers, names, boundaries and lengths.

of this section of the river is its large meanders and the associated muddy shoals and areas of shallow water. Pre-survey reconnaissance visits further refined the survey selection by excluding inaccessible or unsuitable habitat, such as canalised sections with no muddy edges or shallow water. The remaining 8.4km of the river was split into eight sectors of various lengths, defined by access points and physical or geographical obstacles (Table 1, p.55).

A team of 20 volunteers was recruited to manage and undertake the survey, via articles published in Glamorgan Bird Club and local BTO regional newsletters. The Ely Valley Green Sandpiper Study Group was established in 2020.

Monthly coordinated counts were made between October and March, on dates coinciding with WeBS Core Counts dates. For consistency and to minimise the risk of counts being affected by human disturbance (e.g. dog walkers), counts always began thirty minutes after sunrise and were undertaken regardless of weather conditions or river levels. In order to reduce double-counting and increase the likelihood of recording all the birds present, volunteers where possible worked in teams of two or more. Observers were asked to record the following:

- The start and finish time of each count
- Weather/temperature
- Basic assessment of the river level (low/average/high)
- Exact time and location (10-figure OS grid reference) for each sighting
- The direction in which each bird flushed (upstream/downstream/away from the river)
- Whether flushed birds were seen to land within the sector or, if not, whether they were last seen flying in the direction of another sector. NB This information was relayed in real time during each count to all other team members in the field, across all sectors, via WhatsApp
- An estimate of the minimum 'safe' number and possible maximum number of Green Sandpipers present in their sector during each count.



Green Sandpiper on the River Ely: Tim Collier

Data processing

On completion of each monthly count, every sighting was cross referenced with all other sightings, including the timing and direction of flight for flushed birds in order to create a picture of any potential movements of birds between count sectors. Likely double-counts were excluded from the 'safe' minimum count and any birds we were unsure whether they'd been previously counted were recorded in our possible maximum count.

Population Modelling

Population modelling was applied to the field survey count data, in order to improve understanding of the population present at the site and compensate for any remaining double- or under-counting of these mobile birds. Using the field survey datasets, two separate predictive count models were developed from the minimum safe counts and the maximum counts. These used a Generalised Linear Mixed Model (GLMM) with a Bayesian framework, run in the R package. (Brooks *et al* 2017; R Core Team, 2022). Models were fitted with a Poisson error distribution. Each model was fitted with four Markov-Chain Monte Carlo (MCMC) chains, a method that draws samples where the next sample is dependent on the existing sample, called a Markov Chain. Each MCMC chain had 6,000 iterations (i.e. 2,000 post-warmup samples) in order to achieve satisfactory convergence on a final population estimate. The model fixed effects were month of count and (log-transformed) count sector length. From these two models, posterior distributions were generated for the entire surveyed length of the river, and also for individual count sectors. Mean and 95% confidence intervals were calculated from each posterior distribution.

Results

A total of fifteen counts were undertaken across four winters. The first coordinated monthly counts commenced in November 2020, but Covid-19 restrictions prevented field visits in Jan-Mar 2021. Monthly counts resumed in October 2021 and continued to March 2022; recommencing in October 2022 to March 2023. An additional coordinated count was held in January 2024 and it is the group's intention to continue making an annual count in January for the foreseeable future. All counts were completed on the target dates except for 11 December 2022, which was postponed for a week because of dangerous road conditions during a period of very cold weather.

At least one Green Sandpiper was recorded in every sector surveyed on at least one occasion. Sectors 2 and 6 (Table 1) had no Green Sandpiper records prior to this study. The highest 'minimum safe' count was 15 birds in January 2023, although three counts had maximum counts of 18 birds. The lowest minimum counts were seven birds in March 2022 and again in December 2022 (Figure 2). The highest 'minimum safe' count in January 2023 represents an average of one bird in every 0.56km of river surveyed. The modelled population estimates were broadly in agreement with the minimum safe and maximum counts calculated for each sector. (Figure 2 (p.58), Tables 3 (pp.64-66) and 4 (pp.67-69)).

Discussion

Study methodology

Green Sandpipers are extremely shy and flush immediately on seeing an observer, often at long distance (pers. obs), and this trait has been noted by other observers (e.g. Lack 1986; Hayman *et al.* 1987). Sometimes they will flush high into the air, calling noisily; they may land at great distance, or continue out of sight. They may also flush quietly, low to the water, and disappear around the next bend in the river and these birds can easily be missed by observers. Both behaviours present a challenge to obtaining accurate counts, since the first behaviour risks double-counting and the second under-counting if the departure is missed. Given the potential importance of accurate counts for understanding winter populations, our first priority was to attempt to minimise under-recording at the

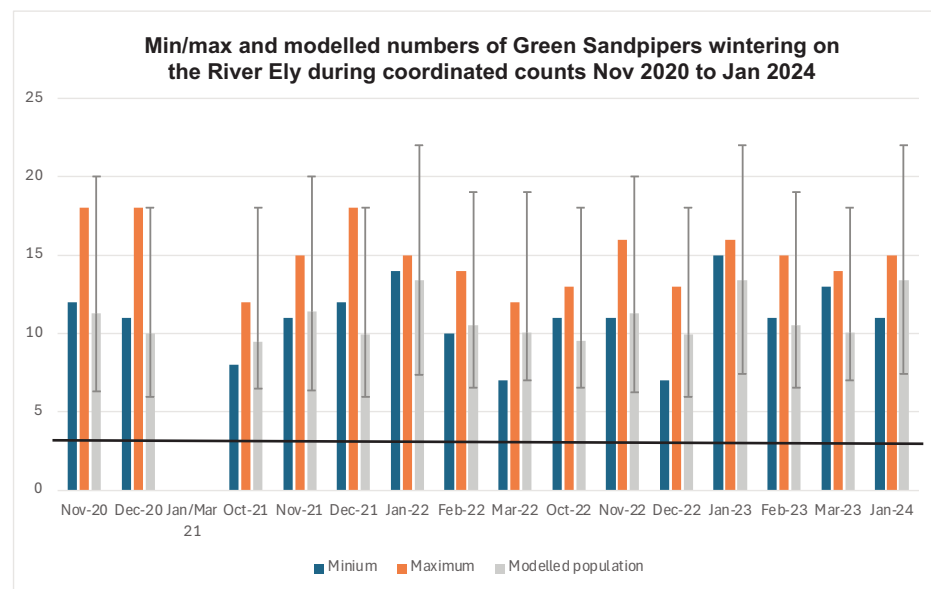


Figure 2. 'Minimum safe', potential maximum and modelled numbers of Green Sandpipers on the River Ely during coordinated counts: Nov/Dec 2020; Oct 2021 to Mar 2022; Oct 2022 to Mar 2023 and Jan 2024. Error bars on the modelled counts represent 95% confidence intervals. The black line represents 1% of the estimated UK wintering population (Frost *et al.* 2019).

same time as eliminating 'double-counting'. Multiple observers working as a team will somewhat increase disturbance, but at the same time increase the detection of quiet birds; whilst real-time communication between teams and careful behaviour recording helped the count processing team exclude likely double-counts. This cautious approach means that it is possible for some records to have been excluded from the 'safe minimum' count in error, but such birds would still be included in the 'likely maximum' count. Some confidence in the ability of the survey method in compensating for flushing behaviour is provided by the results of the population modelling, which is designed to analytically compensate for under- and over-counting of mobile species. While some caution is needed in interpreting the results due to the small sample sizes, it does suggest that the "minimum safe" and "maximum count" approach provided robust estimates of the wintering population of Green Sandpiper for this study site. It may also be worth noting that the lower confidence intervals for each predicted estimate were above the 1% (three bird) threshold.

Habitat as a survey targeting tool

Sectors 2 and 6 were targeted by the survey planning team, despite few previous visits from birdwatchers and no historic Green Sandpiper records. A maximum of two birds were recorded on both these sectors during the survey. This suggests that potentially suitable habitat can be identified using a combination of remote observation and local knowledge, which might be a useful tool for observers wishing to survey Green Sandpipers in other sites, or for improving survey accuracy and coverage at known sites. During the planning period, we searched for stretches of river which had low or medium low banks, meanders and areas of shallow water and muddy shoals which would provide plenty of feeding opportunities. We did not prioritise surveying the small number of canalised stretches of the Ely in our study area where the river tends to run deeper and lacks any muddy edges. During our study, we also found that Green Sandpipers were less often present in areas with high vegetation levels along

the riverbank and were absent from well-wooded riverbanks. We presume that such areas are less attractive because they prevent the birds from having an early view of an approaching threat.

Seasonal variation in bird counts

During the first complete winter of counts (2021-22) the overall Green Sandpiper counts increased each month from October, peaking in January. Thereafter, numbers fell in successive months as spring approached. One study of colour-ringed Green Sandpipers wintering in southern England between 1982 and 1991 found that birds moved from areas of still water to running water as winter progressed (Smith *et al.* 1992). The authors of this study suggest that this might be related to prey availability. The usually drier months of late summer and autumn mean that areas of still water have shallow, muddy fringes offering plenty of feeding opportunities. As winter progresses and water levels rise these areas are lost, while at the same time, vegetation dieback and clearance work increase feeding opportunities along the margins of flowing waters.

Counts were less predictable during the second winter (2022-23), when the December count was the joint lowest count of the study. However, we note that this count was made at the end of a period of exceptionally cold weather when south east Wales recorded ten consecutive sub-zero nights, dropping to -6.9°C in nearby Cardiff on 16 December. Daytime temperatures on several dates during this period failed to reach positive figures. (Alan Rosney, pers. comm. 2024 and in Jones *et al.* 2024). This meant that ice was present in many of the shallow areas of the river favoured by Green Sandpipers for feeding.

Cold weather notwithstanding, the peak count of that winter's survey again occurred in January. It appears that we observed the same pattern of increasing numbers, so this seasonal shift of feeding opportunity from still waters to riverine habitat offers a plausible explanation for the increase in the number of Green Sandpipers moving into our study area as winter progressed.

Spatial behaviour in winter

There were 'hotspots' in several count sectors where Green Sandpipers were almost always present. In Sectors 4 and 5 at Peterston and Pendoylan Moors these represent the largest meanders in the Ely, but the numbers of birds present in these hotspots varied considerably during the course of a winter. Indeed, numbers varied in all the sectors, with birds sometimes recorded in one month, absent the next, and present a month later. This suggests that the wintering population on the Ely can be very mobile, probably responding to river levels, weather conditions and feeding opportunities. This observation contrasts with findings from a colour-ringing study in (primarily) stillwaters and inland streams in Hertfordshire between 1982 and 1991, where the authors found birds to be extremely site-faithful even between years (Smith *et al.* 1992). The same study also found that, despite often arriving in small flocks, some birds defended territories as winter progressed.

To the best of our knowledge no comparable surveys have previously been conducted for Green Sandpipers wintering on linear features like rivers, and our study represents the first such data. This makes comparison between sites somewhat unreliable, especially given the birds' apparent seasonal shift from still water to river habitat as winter progresses. However, our data suggest that Green Sandpipers are present in higher densities in the Ely valley than in other studied areas which include still waters. For example, the mean density in Ken Smith's study area in Hertfordshire was 0.05 birds per km² (range 0.038-0.065) in an area of 306km² (Smith *et al.* 1992). In another extensive study of wintering Green Sandpiper in the Netherlands, Heg (1988) estimated that the overall density was 0.03-0.05 birds per km² in a study area of 1,500km².

Significance of the Ely Valley wintering population

Looking at all WeBS data between 2016-17 and 2022-23 there were only six double-figure counts of

wintering Green Sandpipers anywhere else in the UK during the core wintering period (Nov-Mar), and all of these were areas of still water or marshes. These counts were recorded at Lake 74 - Cotswold Water Park (10 in March 2017); Beddington Sewage Farm (14 in December 2020 and 10 in December 2021); Coed Bach, Gwendraeth (11 in February 2021), the Somerset Levels (16 in December 2021) and at Chew Valley Lake (11 in November 2022). None of these sites recorded double figure counts on more than one occasion in a single winter during this period. During our study in 2020-24, the Ely valley recorded double figure counts of wintering birds on twelve of our fourteen coordinated counts during the core Nov-Mar period across four winters, averaging just over eleven birds each month. This means that, based on the most recent population estimate in Frost *et al.* 2019 and Pritchard *et al.* 2021, the Ely valley held up to 5% of the UK's and 10-15% of Wales' wintering Green Sandpiper population during our study period.

Given the numbers of Green Sandpiper found in this study, and their difficult and contrasting flushing behaviour, it is possible that the current wintering population estimate represents a national underestimate of the species (Frost pers.com). The methods employed in this study might be applied more widely to improve the national estimate, or for local sites to improve confidence in wintering Green Sandpiper numbers in future.

The future of the Ely Valley population

Although the Ely Valley appears to exceed the '1% of population' qualifying threshold for designation for Green Sandpiper, it does not achieve the minimum qualifying threshold which is set at 50 individuals for most species, regardless of population size. (Stroud *et al.* 2014). For the purpose of comparison, Cotswold Water Park was designated as a SSSI by Natural England in 2021 for its rich array of waterbirds and aquatic plants, and Green Sandpiper is noted as one of its features (Anthony 2022), based on a five-year 'peak average' population of 12 birds (DEFRA 2021). The authors note that the Ely Valley study area is already a SSSI, designated for the flowering plant, Monkshood *Aconitum napellus*. Given the significant and possibly increasing wintering population in the Ely Valley, it might seem appropriate for Green Sandpiper to be added as a feature of this existing SSSI by Natural Resources Wales (NRW), although this is not possible under existing legislation without a complete renotification of the site. This perhaps highlights a weakness in our current wildlife legislation. Notification is a costly and lengthy process, and it seems unlikely such action will be undertaken at Ely Valley. The authors hope that this study will highlight the importance of the wintering population of Green Sandpiper to NRW, and consider its potential to be a "qualifying feature" of the Ely Valley SSSI.

Regardless of the lack of designation, the presence of such a significant Green Sandpiper population should be considered, and their wellbeing safeguarded, if any developments or work are proposed or undertaken in the valley. We consider the following to be the most likely potential threats to Green Sandpiper wintering in the Ely valley:

- **Urban expansion and associated disturbance.** The Ely is a densely populated catchment, home to 170,000 people, 5% of the Welsh population. (<https://afonyddcymru.org/wp-content/uploads/2023/10/Ely-3.pdf>). There are currently several building proposals close to the study area and, historically, a plan to build a road linking Cardiff Airport with the M4 which would have cut right through the valley. This risk has now passed but it could potentially resurface.
- **Pollution:** agricultural, sewage, air (nitrates) and upstream urban run-off affecting water quality and invertebrate populations. Under the Water Framework Directive, the health of the River Ely is described by Natural Resources Wales as "Poor" (<https://waterwatchwales.naturalresourceswales.gov.uk/en/>) and, in the view of the South East Wales Rivers Trust, it is one of the most polluted rivers in Wales ([https://afonyddcymru.org/wp-](https://afonyddcymru.org/wp-content/uploads/2023/10/Ely-3.pdf)

content/uploads/2023/10/Ely-3.pdf). Fortunately, the freshwater shrimp, *Gammarus*, which casual observations by several members of the group suggest is the primary prey of Green Sandpipers on the river, is tolerant of water pollution (Shahid *et al.* 2017; Hobrough 1973) and can be abundant and easily visible on the water's edge.

- **Climate change.** Changes in rainfall are already resulting in periods of both extremely low flow, exacerbating existing issues with pollution, and flash flooding, both of which could restrict feeding opportunities on the river. Higher water temperatures in future may also negatively impact on waterbirds, including Green Sandpipers', invertebrate food sources (Durance *et al.* 2009).
- **Invasive Non-native Species (INNS) encroachment (and associated management activity).** Himalayan Balsam (*Impatiens glandulifera*) and Japanese Knotweed (*Reynoutria japonica*) are already present on the Ely (Hudson 2023). We noted that the birds avoid areas with large amounts of Balsam or Knotweed. From late spring to early winter these plants form dense banks of vegetation which reach to the water's edge. This will reduce the availability of feeding areas (muddy shoals and fringes) and also restricts their view of potential threats, resulting in loss and degradation of habitat quality. Removing INNS along the riverbank, although welcome and necessary, also poses a risk of disturbing these highly sensitive birds. Consideration should be given to the timing of any proposed INNS removal so that it coincides with those months (May/June) when the birds are largely absent from the river.
- **Tree planting.** This will have similar impacts to the INNS, i.e. loss or degradation of feeding habitat on the river. With the new impetus to plant trees to help combat the climate emergency, care should be given not to plant trees along those stretches of riverbank that offer the best feeding opportunities for the birds.

In any future discussions about the conservation of the Green Sandpipers in the Ely valley, it should be remembered that this is not a transient population of birds. Some individuals will be passing through on the way to their wintering/breeding grounds, but there are Green Sandpipers present in the valley almost throughout the year, departing only for two or three months to breed. Several studies have revealed a high degree of site fidelity by this species. These range from a return rate from one winter to the next of 56% at the sewage farms of Munster, northern Germany (OAG Munster.1989) to 83.5% in Hertfordshire (Smith *et al.* 1992). This suggests that most of the birds wintering in the Ely valley are ones that return there year after year and that they are highly dependent on the valley.

We hope our study will encourage others to search for and survey Green Sandpipers in their local area. Our methods for site selection might help improve survey coverage and, consequently, the accuracy of the UK and Welsh wintering population estimates. This may be important at wider spatial scales than the UK, since range edges are most sensitive to large-scale pressures such as climate change and this species is at the very north western edge of its wintering range here in Wales.

Further coordinated counts over several winters and an effort to colour-ring the birds may help provide a better assessment of the value, significance and use of the Ely valley by wintering Green Sandpipers. However, we believe that the historic data and the results of our study to date suggest that the Ely does indeed hold a population of national importance of this attractive wader, at both a Welsh and UK level.

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Ethical Statement

No part of this study required the capture or handling of wild birds, or any other licensable activity.

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Sector	Month	Year	log_length	Prediction from min count	LCI	Median	UCI
1	November	2020	0.392042088	2.008916667	0	2	5
2	November	2020	-0.820980552	0.6055	0	0	3
3	November	2020	0.122217633	1.530166667	0	1	4
4	November	2020	-0.116533816	1.21475	0	1	4
5	November	2020	0.832909123	3.099666667	0	3	7
6	November	2020	-0.127833372	1.179	0	1	4
7	November	2020	-0.274436846	1.034833333	0	1	3
8	November	2020	-0.653926467	0.689166667	0	0	3
November Total				11.362			
1	December	2020	0.392042088	1.750333333	0	2	5
2	December	2020	-0.820980552	0.512916667	0	0	2
3	December	2020	0.122217633	1.320166667	0	1	4
4	December	2020	-0.116533816	1.058166667	0	1	4
5	December	2020	0.832909123	2.709416667	0	3	7
6	December	2020	-0.127833372	1.035083333	0	1	3
7	December	2020	-0.274436846	0.892416667	0	1	3
8	December	2020	-0.653926467	0.630416667	0	0	3
December Total				9.908916668			
1	October	2021	0.392042088	1.67925	0	1	5
2	October	2021	-0.820980552	0.499666667	0	0	2
3	October	2021	0.122217633	1.27875	0	1	4
4	October	2021	-0.116533816	1.008833333	0	1	3
5	October	2021	0.832909123	2.607	0	2	7
6	October	2021	-0.127833372	0.99175	0	1	3
7	October	2021	-0.274436846	0.863083333	0	1	3
8	October	2021	-0.653926467	0.586333333	0	0	2
October Total				9.514666666			
1	November	2021	0.392042088	1.993166667	0	2	5
2	November	2021	-0.820980552	0.605083333	0	0	2
3	November	2021	0.122217633	1.52925	0	1	4
4	November	2021	-0.116533816	1.211666667	0	1	4
5	November	2021	0.832909123	3.0995	0	3	7
6	November	2021	-0.127833372	1.1895	0	1	4
7	November	2021	-0.274436846	1.024833333	0	1	3
8	November	2021	-0.653926467	0.688666667	0	0	3
November Total				11.34166667			
1	December	2021	0.392042088	1.752	0	2	5
2	December	2021	-0.820980552	0.513333333	0	0	2
3	December	2021	0.122217633	1.322	0	1	4
4	December	2021	-0.116533816	1.04675	0	1	4
5	December	2021	0.832909123	2.719833333	0	3	7
6	December	2021	-0.127833372	1.046	0	1	4
7	December	2021	-0.274436846	0.913916667	0	1	3
8	December	2021	-0.653926467	0.6125	0	0	2
December Total				9.926333333			
1	January	2022	0.392042088	2.355666667	0	2	6
2	January	2022	-0.820980552	0.71075	0	1	3
3	January	2022	0.122217633	1.8005	0	2	5
4	January	2022	-0.116533816	1.41525	0	1	4
5	January	2022	0.832909123	3.65525	0	3	8

Sector	Month	Year	log_length	Prediction from min count	LCI	Median	UCI
6	January	2022	-0.127833372	1.409583333	0	1	4
7	January	2022	-0.274436846	1.21225	0	1	4
8	January	2022	-0.653926467	0.825	0	1	3
January Total				13.38425			
1	February	2022	0.392042088	1.856833333	0	2	5
2	February	2022	-0.820980552	0.54525	0	0	2
3	February	2022	0.122217633	1.422583333	0	1	4
4	February	2022	-0.116533816	1.11025	0	1	4
5	February	2022	0.832909123	2.903416667	0	3	7
6	February	2022	-0.127833372	1.106333333	0	1	4
7	February	2022	-0.274436846	0.952833333	0	1	3
8	February	2022	-0.653926467	0.640583333	0	0	3
February Total				10.53808333			
1	March	2022	0.392042088	1.75575	0	2	5
2	March	2022	-0.820980552	0.52475	0	0	2
3	March	2022	0.122217633	1.363166667	0	1	4
4	March	2022	-0.116533816	1.066333333	0	1	4
5	March	2022	0.832909123	2.745583333	0	3	7
6	March	2022	-0.127833372	1.051083333	0	1	4
7	March	2022	-0.274436846	0.897416667	0	1	3
8	March	2022	-0.653926467	0.624	0	0	3
March Total				10.02808333			
1	October	2022	0.392042088	1.683166667	0	1	5
2	October	2022	-0.820980552	0.505166667	0	0	2
3	October	2022	0.122217633	1.273333333	0	1	4
4	October	2022	-0.116533816	1.005833333	0	1	3
5	October	2022	0.832909123	2.6055	0	2	6
6	October	2022	-0.127833372	0.982416667	0	1	3
7	October	2022	-0.274436846	0.858833333	0	1	3
8	October	2022	-0.653926467	0.598333333	0	0	2
October Total				9.512583333			
1	November	2022	0.392042088	1.993083333	0	2	5
2	November	2022	-0.820980552	0.604166667	0	0	2
3	November	2022	0.122217633	1.526666667	0	1	4
4	November	2022	-0.116533816	1.192333333	0	1	4
5	November	2022	0.832909123	3.113083333	0	3	7
6	November	2022	-0.127833372	1.19225	0	1	4
7	November	2022	-0.274436846	1.029666667	0	1	3
8	November	2022	-0.653926467	0.709833333	0	1	3
November				11.36108333			
1	December	2022	0.392042088	1.763666667	0	2	5
2	December	2022	-0.820980552	0.52525	0	0	2
3	December	2022	0.122217633	1.332416667	0	1	4
4	December	2022	-0.116533816	1.064833333	0	1	4
5	December	2022	0.832909123	2.718583333	0	3	7
6	December	2022	-0.127833372	1.032	0	1	3
7	December	2022	-0.274436846	0.893833333	0	1	3

Sector	Month	Year	log_length	Prediction from min count	LCI	Median	UCI
8	December	2022	-0.653926467	0.6135	0	0	3
	December Total			9.944083333			
1	January	2023	0.392042088	2.341	0	2	6
2	January	2023	-0.820980552	0.707	0	1	3
3	January	2023	0.122217633	1.809	0	2	5
4	January	2023	-0.116533816	1.403083333	0	1	4
5	January	2023	0.832909123	3.634833333	0	3	8
6	January	2023	-0.127833372	1.400333333	0	1	4
7	January	2023	-0.274436846	1.220666667	0	1	4
8	January	2023	-0.653926467	0.820833333	0	1	3
	January Total			13.33675			
1	February	2023	0.392042088	1.869416667	0	2	5
2	February	2023	-0.820980552	0.552416667	0	0	2
3	February	2023	0.122217633	1.414916667	0	1	4
4	February	2023	-0.116533816	1.1255	0	1	4
5	February	2023	0.832909123	2.89125	0	3	7
6	February	2023	-0.127833372	1.103083333	0	1	4
7	February	2023	-0.274436846	0.9465	0	1	3
8	February	2023	-0.653926467	0.667333333	0	0	3
	February Total			10.57041667			
1	March	2023	0.392042088	1.78275	0	2	5
2	March	2023	-0.820980552	0.512416667	0	0	2
3	March	2023	0.122217633	1.3465	0	1	4
4	March	2023	-0.116533816	1.057416667	0	1	4
5	March	2023	0.832909123	2.731833333	0	3	7
6	March	2023	-0.127833372	1.055833333	0	1	4
7	March	2023	-0.274436846	0.898583333	0	1	3
8	March	2023	-0.653926467	0.62125	0	0	3
	March Total			10.00658333			
1	January	2024	0.392042088	2.35525	0	2	6
2	January	2024	-0.820980552	0.6955	0	1	3
3	January	2024	0.122217633	1.788583333	0	2	5
4	January	2024	-0.116533816	1.410416667	0	1	4
5	January	2024	0.832909123	3.66875	0	3	8
6	January	2024	-0.127833372	1.403416667	0	1	4
7	January	2024	-0.274436846	1.2095	0	1	4
8	January	2024	-0.653926467	0.823333333	0	1	3
	January Total			13.35475			

Table 3. Minimum safe number of green sandpiper predicted for each sector per month.

Sector	month	year	log_length	Prediction from max count	LCI	Median	UCI
1	November	2020	0.39	2.86	0	3	7
2	November	2020	-0.82	0.87	0	1	3
3	November	2020	0.12	2.22	0	2	6
4	November	2020	-0.12	1.74	0	2	5
5	November	2020	0.83	4.49	1	4	9
6	November	2020	-0.13	1.71	0	2	5
7	November	2020	-0.27	1.49	0	1	4
8	November	2020	-0.65	1.03	0	1	3
	November			16.43			
1	December	2020	0.39	2.88	0	3	7
2	December	2020	-0.82	0.86	0	1	3
3	December	2020	0.12	2.19	0	2	6
4	December	2020	-0.12	1.73	0	2	5
5	December	2020	0.83	4.43	1	4	9
6	December	2020	-0.13	1.72	0	2	5
7	December	2020	-0.27	1.46	0	1	4
8	December	2020	-0.65	1.02	0	1	3
	December Total			16.28			
1	October	2021	0.39	2.22	0	2	6
2	October	2021	-0.82	0.65	0	0	3
3	October	2021	0.12	1.68	0	1	5
4	October	2021	-0.12	1.31	0	1	4
5	October	2021	0.83	3.42	0	3	8
6	October	2021	-0.13	1.31	0	1	4
7	October	2021	-0.27	1.12	0	1	4
8	October	2021	-0.65	0.78	0	1	3
	October Total			12.50			
1	November	2021	0.39	2.90	0	3	7
2	November	2021	-0.82	0.85	0	1	3
3	November	2021	0.12	2.21	0	2	6
4	November	2021	-0.12	1.70	0	2	5
5	November	2021	0.83	4.44	1	4	9
6	November	2021	-0.13	1.72	0	2	5
7	November	2021	-0.27	1.47	0	1	4
8	November	2021	-0.65	1.00	0	1	3
	November			16.30			
1	December	2021	0.39	2.89	0	3	7
2	December	2021	-0.82	0.84	0	1	3
3	December	2021	0.12	2.21	0	2	6
4	December	2021	-0.12	1.75	0	2	5
5	December	2021	0.83	4.45	1	4	9
6	December	2021	-0.13	1.70	0	2	5
7	December	2021	-0.27	1.47	0	1	4
8	December	2021	-0.65	0.99	0	1	3
	December Total			16.29			
1	January	2022	0.39	2.72	0	3	7
2	January	2022	-0.82	0.80	0	1	3
3	January	2022	0.12	2.05	0	2	5

Sector	month	year	log_length	Prediction from max count	LCI	Median	UCI
4	January	2022	-0.12	1.63	0	1	5
5	January	2022	0.83	4.21	1	4	9
6	January	2022	-0.13	1.61	0	1	5
7	January	2022	-0.27	1.39	0	1	4
8	January	2022	-0.65	0.96	0	1	3
	January Total			15.38			
1	February	2022	0.39	2.54	0	2	6
2	February	2022	-0.82	0.75	0	1	3
3	February	2022	0.12	1.95	0	2	5
4	February	2022	-0.12	1.53	0	1	5
5	February	2022	0.83	3.95	1	4	9
6	February	2022	-0.13	1.51	0	1	4
7	February	2022	-0.27	1.31	0	1	4
8	February	2022	-0.65	0.90	0	1	3
	February Total			14.45			
1	March	2022	0.39	2.29	0	2	6
2	March	2022	-0.82	0.68	0	0	3
3	March	2022	0.12	1.74	0	2	5
4	March	2022	-0.12	1.37	0	1	4
5	March	2022	0.83	3.56	0	3	8
6	March	2022	-0.13	1.37	0	1	4
7	March	2022	-0.27	1.18	0	1	4
8	March	2022	-0.65	0.79	0	1	3
	March Total			12.97			
1	October	2022	0.39	2.19	0	2	6
2	October	2022	-0.82	0.65	0	0	3
3	October	2022	0.12	1.68	0	1	5
4	October	2022	-0.12	1.33	0	1	4
5	October	2022	0.83	3.43	0	3	8
6	October	2022	-0.13	1.32	0	1	4
7	October	2022	-0.27	1.14	0	1	4
8	October	2022	-0.65	0.79	0	1	3
	October Total			12.53			
1	November	2022	0.39	2.86	0	3	7
2	November	2022	-0.82	0.85	0	1	3
3	November	2022	0.12	2.20	0	2	6
4	November	2022	-0.12	1.73	0	2	5
5	November	2022	0.83	4.46	1	4	9
6	November	2022	-0.13	1.69	0	1	5
7	November	2022	-0.27	1.45	0	1	4
8	November	2022	-0.65	1.01	0	1	3
	November Total			16.25			
1	December	2022	0.39	2.88	0	3	7
2	December	2022	-0.82	0.86	0	1	3
3	December	2022	0.12	2.21	0	2	6
4	December	2022	-0.12	1.72	0	2	5
5	December	2022	0.83	4.50	1	4	9
6	December	2022	-0.13	1.71	0	2	5

Sector	month	year	log_length	Prediction from max count	LCI	Median	UCI
7	December	2022	-0.27	1.48	0	1	4
8	December	2022	-0.65	1.01	0	1	3
	December Total			16.38			
1	January	2023	0.39	2.70	0	3	6
2	January	2023	-0.82	0.80	0	1	3
3	January	2023	0.12	2.06	0	2	5
4	January	2023	-0.12	1.63	0	1	5
5	January	2023	0.83	4.18	1	4	9
6	January	2023	-0.13	1.63	0	1	5
7	January	2023	-0.27	1.39	0	1	4
8	January	2023	-0.65	0.98	0	1	3
	January Total			15.37			
1	February	2023	0.39	2.54	0	2	6
2	February	2023	-0.82	0.77	0	1	3
3	February	2023	0.12	1.96	0	2	5
4	February	2023	-0.12	1.53	0	1	4
5	February	2023	0.83	3.96	1	4	9
6	February	2023	-0.13	1.52	0	1	4
7	February	2023	-0.27	1.33	0	1	4
8	February	2023	-0.65	0.90	0	1	3
	February Total			14.52			
1	March	2023	0.39	2.26	0	2	6
2	March	2023	-0.82	0.69	0	0	3
3	March	2023	0.12	1.73	0	2	5
4	March	2023	-0.12	1.38	0	1	4
5	March	2023	0.83	3.54	0	3	8
6	March	2023	-0.13	1.36	0	1	4
7	March	2023	-0.27	1.17	0	1	4
8	March	2023	-0.65	0.81	0	1	3
	March Total			12.95			
1	January	2024	0.39	2.74	0	3	7
2	January	2024	-0.82	0.80	0	1	3
3	January	2024	0.12	2.06	0	2	5
4	January	2024	-0.12	1.63	0	1	5
5	January	2024	0.83	4.19	1	4	9
6	January	2024	-0.13	1.61	0	1	5
7	January	2024	-0.27	1.37	0	1	4
8	January	2024	-0.65	0.94	0	1	3
	January Total			15.34			

Table 4. Maximum number of green sandpiper predicted for each sector per month. These numbers are predicted from a GLMM model created using maximum counts. 95% upper and lower confidence intervals are shown.

Museum archives reveal artwork likely to depict the last breeding pair of White-tailed Eagles in South Wales

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Crynodeb

Mae Eryr y Môr yn rhan hanesyddol o ecoleg a diwylliant Cymru, fel aderyn nythu y credir iddo ddiflannu tua chanol y ddeunawfed ganrif oherwydd erledigaeth ddynol. Yma, yr ydym yn disgrifio darlun dyfrlliw yn archif Amgueddfa Cymru ac yn trafod ei oblygiadau ar gyfer ein dealltwriaeth o hanes Eryr y Môr yng Nghymru. Mae’r darlun gan William Weston Young, sy’n dyddio i 1816, yn dangos efallai y pâr olaf o Eryrod y Môr i nythu yn ne Cymru, gyda’r iâr wedi ei dal mewn trap a’r ceiliog yn hofran uwchben. Disgrifiwyd yr erledigaeth ar y ddau aelod o’r pâr mewn newyddiadur lleol yn 1828. Mae’r cofnodion hanesyddol yma felly yn disgrifio efallai y pâr olaf o Eryrod y Môr i nythu yn ne Cymru, cyn i’r rhywogaeth gael ei difa.

Abstract

White-tailed Eagles, or Eryr y Môr in Welsh, are a historic part of Wales’ biodiversity, heritage and culture, as a breeding bird of prey that is thought to have gone extinct in the mid-18th century due to human persecution. We here describe a watercolour illustration in the Amgueddfa Cymru – Museum Wales archives and discuss its implications for our understanding of the history of White-tailed Eagles in Wales. The painting by William Weston Young, dated 1816, depicts a female near Kenfig (Bridgend County Borough) caught in a trap and a male hovering above. Persecution of both eagles was later documented in a local newspaper in 1828. The described behaviours are consistent with the two birds having formed a residential pair at or near Kenfig. These historical records hence document what may have been the last – possibly breeding – pair of White-tailed Eagles in South Wales, before the species was driven to extinction.

Introduction

The White-tailed Eagle, *Haliaeetus albicilla*, historically bred in Wales, and became extinct as a breeding species the mid-1800s. Population estimates of pairs present across Britain and Ireland plummeted, from 2,550 in 5000 years before present, to 50 around 1850, then 0 by 1920, based on historical records (Evans *et al.* 2012). The species’ breeding ceased in England by the 1780s, and later in Scotland, with the last bird being killed in Shetland in 1918 (Yalden 2007; Evans *et al.* 2012). These local extinctions on the British Isles were driven by human persecution (Love, 1983). However, there has since been a tremendous effort to restore White-tailed Eagles to Britain and Ireland, with the first successful reintroduction being on the Isle of Rum National Nature Reserve in 1975 (Love 1988). After the successes in Western Scotland, White-tailed Eagles were later also reintroduced to eastern Scotland (RSPB 2013) and Ireland (Nygård *et al.* 2009, Mee *et al.* 2016) in 2007, and more recently to the Isle of Wight in 2019 (Dennis *et al.* 2019). White-tailed Eagles have not yet been restored to Wales, although there is an ongoing evaluation of feasibility for reintroduction, driven by Eagle Reintroduction Wales (<https://www.eaglereintroductionwales.com>). IUCN guidelines suggest that reintroductions should

only be carried out where there is evidence of the species historically occurring as a breeding species within the proposed reintroduction area (IUCN/SSC 2013). Therefore, it is paramount to understand the historical distribution of the White-tailed Eagle as a native breeding species across the British Isles and specifically the Welsh landscape.

The White-tailed Eagle has historically been referred to in English as the Sea Eagle, Erne, Great Erne, and in Welsh as Eryr y Môr (Williams *et al.* 2020). Studies of historical observations, place names and archaeological records have identified former occurrence of White-tailed Eagles across Wales (Yalden 2007; Evans *et al.* 2012; Lewis 2021; Williams *et al.* 2020; Pritchard 2020). Williams *et al.* (2020) identified that all modern counties in Wales had historical records of White-tailed Eagles, with historic records being more abundant in areas of Gwynedd, Dyfed, West Glamorgan and Mid Glamorgan. It should be noted, however, that Lewis (2021) took a much more cautious approach, concluding that based on ornithological literature there is little or no evidence of the species being common or breeding in Wales historically.

Williams *et al.* (2020) identified that White-tailed Eagles were extinct as a breeding species in Wales by the early 1800s, but during the 19th century White-tailed Eagles were frequently observed in Gwynedd and around Kenfig, Bridgend and Margam. The Kenfig area was responsible for over 50 years of records, between 1810 and 1860, and one later record, in 1906 (Williams *et al.* 2020). Two individuals which were possibly the last pair of White-tailed Eagles in the Kenfig area were persecuted, with the female shot in 1816, and the male shot 1828, which was reported in *The Carmarthen Journal* and *South Wales Weekly Advertiser* in 1828 (Welsh Online Newspaper Library 2024, Williams *et al.* 2020). The newspaper article reports a ‘Great Erne’ was shot by a principal game keeper, and reports morphometrics of the bird including it weighing more than 10 pounds [4.54 kg], 3 feet [91.4 cm] from head to end of tail, seven feet four inches [223.5 cm] from wing tip to wing tip extended, bill three inches [7.6 cm] in length. The newspaper article suggests that the male was the mate of the female Erne caught in Kenfig in 1816, as an eagle had been observed at various intervals in the area, highlighting the longevity of the species. White-tailed Eagles exhibit strong natal philopatry and commitment to natal and breeding sites, therefore it is not surprising there is a record of a male eagle present in the area, long after its presumed mate was persecuted at Kenfig. However, we cannot fully rule out that the recorded male eagle could have been a different individual.

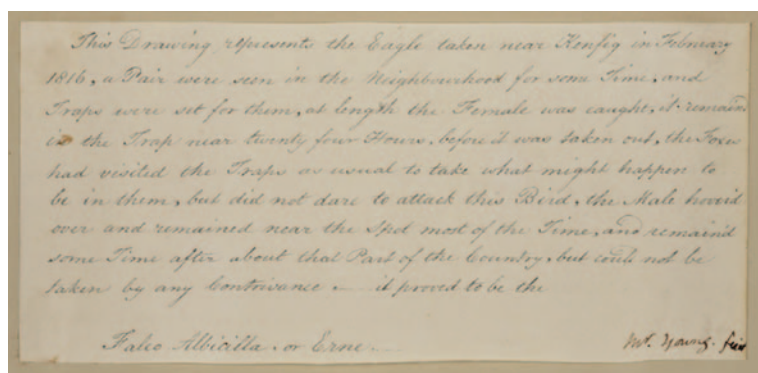
Further evidence regarding these two eagles described in the newspaper article was discovered by Dewi Lewis (2021) in the Amgueddfa Cymru – Museum Wales artwork archive. The archive is home to a watercolour painting dated February 1816, of a female White-tailed Eagle caught in a trap, with a male flying overhead, and a Red Fox (*Vulpes vulpes*) waiting in the distance. We here reproduce the painting, with kind permission from Amgueddfa Cymru (Figure 1, p.72). The artwork was painted by William Weston Young (1776–1847), among other trades an artist, botanist and entrepreneur, and accompanied by a transcript detailing the events of the painting. The transcript (p.72) relays that the trap had been set for some time, until the female was caught and remained in the trap for 24 hours. The female in the painting likely refers to the female shot in Kenfig in 1816, reported in the local newspaper (Welsh Online Newspaper Library 2024, Williams *et al.* 2020). The artwork was previously reported by Lewis (2021), though the newspaper article associating the eagles with records of persecution was not linked to the evidence from this painting.

The cross-referencing of historical material can provide stronger evidence of past events; in this case the combination of the artwork with the local newspaper article. Although we cannot be fully certain that the male and female White-tailed Eagles documented in the artwork and newspaper were a resident breeding pair, and indeed quite possibly the last, in South Wales, we believe that this combined evidence provides strong support for this theory. The behaviour of the male remaining associated with



A

Figure 1. (A) Historical water colour painting and (B) inscription of White-tailed Eagle, dated 1816 in Kenfig. Credits (Amgueddfa Cymru – Museum Wales, any enquiries please email images@museumwales.ac.uk)



B

The transcript reads:

'This drawing represents the Eagle taken near Kenfig in February 1816, a pair were seen in the neighbourhood for some time, and Traps were set for them, at length the female was caught, it remained in the trap for twenty four hours, before it was taken out, the foxes had visited the traps as usual to take what might happen to be in them, but did not dare to attack this bird, the male hovered over and remained near the spot most of the time, and remained some time after about that part of the country but could not be taken by any contrivance...it proved to be the *Falco Albicilla* or Erne.'

the female over such long time periods suggests the two being a pair (Glutz von Blotzheim *et al.* 1971; Dementavičius and Treinys 2009). Furthermore, paired White-tailed Eagles in Scotland and Ireland do not tend to co-migrate to non-breeding areas, suggesting the two to be a resident pair at or near Kenfig.

Museum specimens can provide a wealth of biological and societal information, and they can be used for genetic studies. Future studies at Cardiff University will attempt to genetically characterise extinct Welsh White-tailed Eagles, through analysis of museum specimens. Characterising historical DNA combined with genomic monitoring of extant White-tailed Eagles in Britain and Ireland (reintroduced from stocks originating from Norway), will allow us to determine whether a genetically unique and differentiated population of White-tailed Eagle was lost in Wales when they were persecuted to extinction.

Acknowledgements

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Green Sandpiper on the River Ely: Tim Collier

Modelling population-level impacts of wind farm collision risk on Welsh Red Kites

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Crynodeb
Mae poblogaeth y Barcud Coch yng Nghymru yn cynyddu’n gyflym, yn dilyn gostyngiad hanesyddol oherwydd erledigaeth. Un bygythiad modern i’r Barcud yw tyrbinau gwynt yn achosi marwolaethau oherwydd gwrthdaro, ac ar hyn o bryd mae cynigion am lawer o ffermydd gwynt newydd ar y gweill neu wedi eu datblygu i wahanol lefelau. Defnyddiwyd Dadansoddiad Hyfywra Poblogaeth i fodelu effeithiau posibl datblygiadau ffermydd gwynt presennol ac arfaethedig ar boblogaeth Barcud Cymru. Roedd y modelu yn defnyddio amcangyfrifon poblogaeth yn ôl ardal, gyda chyfuniadau o werthoedd uchaf, isaf a chanolig ar gyfer pob paramedr demograffiaidd. Defnyddiwyd proses samplu gwrthodiad i gynhyrchu set derfynol o fodelau, gyda chyfres o dafl-lwybrau poblogaeth sy’n adlewyrchu’r tueddiadau diweddar a welwyd ym mhoblogaeth y Barcud yng Nghymru. Defnyddiwyd sawl gwahanol senario o farwolaethau yn dilyn gwrthdrawiad a thyrbinau gwynt i asesu’r effaith ar raddfa genedlaethol (Cymru), ar raddfa ranbarthol ac o fewn Ardaloedd Gwarchodaeth Arbennig (AGA) Elenydd-Mallaen a’r Berwyn. Dan y senario sylfaenol (yn cynnwys y tyrbinau gwynt presennol ond dim datblygiadau pellach), roedd pob model yn darogan y byddai’r boblogaeth yn gyson neu’n cynyddu, ar bob graddfa ofodol. Bychan oedd effaith tyrbinau gwynt ychwanegol ar dwf y boblogaeth, er bod hyn dros amser yn arwain at wahaniaethau sylweddol ym maint y boblogaeth. Yn gyffredinol, roedd effeithiau a fodelwyd ar gyfer datblygiadau arfaethedig yn fwy mewn ardaloedd llai gyda phoblogaethau Barcud mwy (e.e.. poblogaethau’r ddwy AGA). Gall y canlyniadau hyn roi cadarnhad nad yw’n debygol y bydd y lefel o ddatblygiad ffermydd gwynt sydd ar y gweill ar hyn o bryd yn atal twf poblogaeth y Barcud yng Nghymru. Fodd bynnag, maent hefyd yn dangos effaith ar faint terfynol y boblogaeth yn y tymor hir a bod angen mwy o ofal wrth drafod datblygiadau sy’n agos i’r ddwy AGA, er mwyn lleihau’r perygl i’r poblogaethau o fewn, ac yn enwedig ar gyron, yr ardaloedd gwarchodaeth hyn.

Abstract
The Welsh Red Kite population is currently undergoing rapid growth, following a historic decline driven by persecution. Wind turbines represent a modern threat to Red Kites, causing additional mortality through collisions, and numerous new wind farms are currently proposed and at various stages of development. We used Population Viability Analysis to model the potential impacts of current and proposed wind farm developments on the Welsh Red Kite population. Modelling used spatially-explicit population estimates, together with combinations of upper, lower and intermediate values for each demographic parameter. A rejection-sampling procedure was applied to retain a final model set, whose range of population trajectories reflected the observed recent population trend of Red Kites in Wales. Different scenarios of wind turbine collision mortality were applied, to assess impacts at the national

(Wales) scale, regional scale, and within the Elenydd-Mallaen and Berwyn Special Protection Areas (SPAs). Under the baseline scenario (including existing wind turbines but no further future developments), all models predicted future population stability or increase, at all spatial scales. Impacts of additional wind turbines on population growth were small, although over time these propagate into substantial differences in population size. In general, modelled impacts of proposed developments were larger at smaller spatial scales with larger Red Kite populations (e.g. the two SPA populations). These results may provide reassurance that currently-proposed levels of wind farm development are unlikely to prevent the continued growth of the Welsh Red Kite population. However, they also highlight impacts on the ultimate size of the Red Kite population in the longer term, and that greater caution is justified for developments in proximity to the two SPAs, in order to minimize risk to the populations within, and particularly around the fringes of, these specifically-protected areas.

Introduction

Wind farms in the Welsh landscape

The climate emergency requires an urgent transition from the consumption of fossil fuels to renewable sources of energy (Ripple *et al.* 2020), but renewable energy generation itself has potential environmental impacts that are important to quantify and, where possible, to mitigate. In Wales, proposed and existing renewable energy schemes include inland hydro-electric generation, tidal barrages, solar panel arrays - and most importantly in terms of total energy production – offshore and onshore wind turbines (Llywodraeth Cymru / Welsh Government 2022). One major concern regarding the rapid proliferation of onshore wind turbines in the Welsh landscape is their potential impact on birds of conservation concern, including birds of prey. Birds are sometimes found to be killed by collisions with the turbine blades, and may also suffer non-lethal effects such as displacement from areas with high densities of wind turbines, leading to impacts on abundance and distribution at the population level (e.g. Bellebaum *et al.* 2013, Schaub 2012). As a result, there is an urgent need to assess the impact of the proposed future increases in wind turbines on populations of birds of prey of conservation concern, such as the Red Kite (*Milvus milvus*).

Historic changes in the Red Kite population in the British Isles, including Wales

The Red Kite was historically widespread across the British Isles, but by the 1930s and 1940s intense persecution had reduced the entire British population to fewer than ten breeding pairs, concentrated in a small area of mid-Wales. Careful conservation organised by a local 'Kite Committee' of conservationists and landowners restored the Welsh population to 100 pairs by 1993. Most birds were descended from a single female that had continued to breed successfully during the population bottleneck (Carter 2007). As a step towards restoring the original breeding range, birds were re-introduced to other parts of the British Isles; in 1989 into the Chilterns (Oxfordshire and Buckinghamshire) and into the Black Isle in Easter Ross, Scotland (Evans and Pienkowski 1991). Successful breeding populations quickly established in both areas. Further releases were begun in Northamptonshire in 1995, central Scotland in 1996, Yorkshire in 1999, Dumfries and Galloway in 2001, north-east England in 2004, Aberdeenshire in 2007 and County Down in 2008. Each of these centres of reintroduction has given rise to a productive breeding group, in some cases benefiting from large-scale provision of food (e.g. Orros and Fellowes 2014, 2015), or the development of a well-established communal roost. Introduced birds and their offspring wander widely across Britain and Ireland but, as yet, pairs have been slow to set up breeding sites distant from the release areas (Balmer *et al.* 2013). Nonetheless, the BTO/JNCC/RSPB Breeding Bird Survey (BBS) has recorded a steep rise in abundance since 1995 (Heywood *et al.* 2023). At the UK national scale, there has been an increase of almost 2,000 % over the last 25 years, and the population continues to increase rapidly. In Wales, the species has recovered to the point that population change can now be reported under the annual Breeding Bird Survey. Results from a transect survey specifically targeting Common Buzzards (*Buteo buteo*) and Red Kites in central southern England, between 2011 and 2016, showed a doubling of population in the

area over that period, broadly in line with the BBS trend (Stevens *et al.* 2020).

Contemporary status and conservation action

In light of this sustained population growth, the Red Kite was moved to the UK Birds of Conservation Concern Green list in 2015 (Eaton *et al.* 2015) and the Welsh Green list in 2022 (Johnstone *et al.* 2022). Nonetheless, a range of conservation issues remain relevant; particularly in the context of two Special Protection Areas in Wales which are designated under Article 4.1 of EC Directive 79/409 on the Conservation of Wild Birds for the protection of breeding Red Kite (an Annex 1 species). For this purpose, the SPA populations are considered by Natural Resources Wales (NRW; and its predecessor, the Countryside Council for Wales, CCW) to include “all birds nesting within the SPA, or within 2 km of [i.e. outside] the boundary” (CCW 2008a). The Elenydd-Mallaen SPA (site code UK9014111) was designated in 1996 for holding 9.3 % of the breeding Red Kite population of Great Britain at that time (JNCC 2015a), and held 34 breeding pairs of Red Kite at the most recent formal assessment (JNCC 2015a). The Berwyn SPA (site code UK9013111) was designated in 1998 for holding 1.2 % of the breeding population of Great Britain (JNCC 2015b), and held 2–3 breeding pairs of Red Kite at the most recent formal assessment (JNCC 2015b).

Artificial feeding of Red Kites can help support populations, and inadvertent food provision through contemporary farming approaches (lambing outdoors - particularly in the uplands, availability of sheep afterbirth and fallen stock as carrion) may have contributed to supporting the remnant Red Kite population through its lowest point during the 20th Century (Roger Buisson, pers. comm.). However, there may be a risk that such food provision may artificially inflate populations to such an extent that if feeding was halted (or farming practices changed), there may be insufficient resources in the local area to continue to sustain the number of birds present (Orros and Fellowes 2014). There is also increasing concern that bird aggregations at artificial food supplies might facilitate the transmission of diseases such as Highly Pathogenic Avian Influenza (HPAI; Toms 2023). Some illegal killing of Red Kites continues, and in northern Scotland the deliberate use of poisoned baits to kill raptors has severely limited the growth of the population (Smart *et al.* 2010, Sansom *et al.* 2016). Poisoning also still occurs in England and Wales (RSPB 2023) and may have slowed the rate of expansion in some areas (Molenaar *et al.* 2017). Anticoagulant rodenticides (rat poisons) have been found in some Red Kite carcasses in concentrations sufficient to be considered a contributory cause of death, but there is no evidence so far to suggest that this might be affecting the population growth rate (Walker *et al.* 2013).

An additional potential threat is posed by an increasing drive to expand renewable energy generation in Wales through the development of onshore wind capacity. Red Kites are known to fall victim to collisions with wind turbines (Schaub 2012), including occasionally in Wales (BSG Ecology 2022; Cross 2023). Modelling suggests that the spatial distribution of wind turbines can significantly shape their impact upon Red Kite populations (Schaub 2012, Bellebaum *et al.* 2013), such that assessing environmental impact of wind power plants collectively at regional scale, rather than on a case-by-case basis, is justified (Schaub 2012). However, at this stage it is unclear what impact the current suite of proposed wind energy developments is likely to have on the continued growth of the Red Kite population in Wales, both at national scale and regionally, and with particular reference to the two designated SPAs.

Background to the modelling project: industry need

A large number of wind farms are currently being proposed in Wales (with details of at least 60 supplied confidentially to this project), especially in mid-Wales. Many of these are sufficiently close to the two SPAs that impacts on the Red Kite populations within and surrounding these SPAs must legally be considered. To date, NRW have requested detailed consideration of impacts on SPA Red Kite populations for proposed wind farms, in some cases where the proposed developments are in excess of 8 km from

SPA boundaries.

To be able to complete robust assessments of the impact of wind farm-related mortality on SPA populations from individual developments, population modelling is typically required. This is specialist work that most commercial consultancies are not equipped to do. Completing this on a site-by-site basis, and often under time constraints in the lead up to Environmental Impact Assessment (and Habitats Regulations Assessment) submission, is likely to result in inconsistent approaches in different applications, to both the modelling itself and the interpretation of findings. A more consistent, strategic and national approach is therefore needed. It is also inevitable that in the absence of an overarching study, and as more wind farm proposals are put forward, NRW will become more concerned about in-combination effects from multiple wind farms proposed by multiple developers on SPA populations, and potentially also about impacts on the wider Welsh Red Kite population. This is likely to result in requests for additional survey work, delays in the planning process, and complicated hearings. There is also the clear potential for proposed schemes to be turned down based on the precautionary principle, as Planning and Environmental Decisions Wales (PEDW) place a great deal of weight on NRW's views.

It follows that, to understand the capacity of Red Kite populations to absorb current and future collision fatalities, and thereby give NRW the evidence-base to provide a more informed view on the impacts of wind farm proposals on the SPAs, a strategic study is required. An independently-conducted modelling exercise by a collaboration of experienced researchers was considered to be more influential than a consultancy-led piece of work in this regard. A study building an understanding of Red Kite population resilience at local and national scales was therefore both timely and informative.

BSG Ecology, on behalf of several member organisations of RenewableUK Cymru, therefore approached BTO Cymru to commission a population viability analysis (PVA) for the Welsh Red Kite population, for which the Berwyn and Elenydd-Mallaen SPAs are designated. The renewable energy generation industry's objective was to better understand the resilience of the Welsh Red Kite population to additional mortality through collision with wind turbines; and the vulnerability of the SPA populations to development sites outside of, but close or adjacent to, their boundaries. Although new wind farms may not be located within the SPAs (which are generally considered a hard constraint to development), they may be located close to SPA boundaries. There is therefore potential for Red Kite populations within an SPA to be impacted both by mortality of individuals breeding within, but ranging outside, the SPA boundary (for example to forage, both during the breeding season and when they range more widely outside the breeding season), and by mortality of individuals breeding outside the SPA boundary but providing young birds that would potentially recruit into the SPA population.

The rapid increase in the Red Kite population in Wales over recent decades will likely have resulted in far more pairs breeding within and close to the SPAs than were present when the sites were originally designated. Welsh Red Kites are also likely to have higher breeding productivity than has historically been the case, due to recent genetic mixing with birds from the reintroduced populations in England, as well as occasional immigrating Scottish and Irish birds (Skujina 2013). The population may therefore be resilient to a certain amount of increased mortality from wind turbine collisions. Demographic modelling provides a quantitative framework for examining the sensitivity of the Red Kite population to altered survival risk, under a range of impact scenarios and at a range of spatial scales.

Population models: general approach

Population Viability Analysis (PVA) has become a commonly used tool in conservation biology and in the management of threatened or endangered species (Keedwell 2004). PVA is a general term for demographic predictive models which forecast the robustness of a population to scenarios of impact

compared to an unimpacted baseline (Beissinger *et al.* 2006). Robust PVAs are contingent upon sufficient demographic data (e.g. population estimates, measures of breeding success, juvenile and adult survival estimates; Keedwell 2004). Natural England (NE) have developed a PVA modelling framework (Searle *et al.* 2019) as a front-end, interactive web application user interface (Searle *et al.* 2019; https://github.com/naturalengland/Seabird_PVA_Tool) allowing users to set-up, apply and run their own PVA models without the need for access to specific software. This application was developed within the context of modelling mortality of seabirds as a consequence of collisions with offshore wind turbines, but the modelling tool can be used to assess any type of impact that changes the survival or breeding productivity of any avian species (Searle *et al.* 2019). In other words, the generic nature of the tool is such that it can be applied to other groups of birds. The NE seabird modelling tool is available to other users under an Open Government Licence. In order to meet BTO internal standards of replication/ reproducibility and accuracy, BTO staff developed the capacity to use the underpinning tools for the NE web application within the R programming environment (R Core Team 2022) as part of previous work developing PVAs for Great Cormorant *Phalacrocorax carbo* and Goosander *Mergus merganser* (Macgregor *et al.* 2022).

PVAs are parameterised using a range of demographic metrics (e.g. brood size, adult and immature mortality); which need to be evidence-based. In addition to evidence from the published literature (much of which will refer to Red Kite populations outside Wales), two long-term datasets have been collected that contain demographic information on the Welsh population. Firstly, nest records collated by the Welsh Kite Trust over the period 2000–2013 (Welsh Kite Trust 2023) have enabled a detailed understanding of productivity in the population, and can permit the estimation of breeding “skip rate” (i.e. the likelihood of an adult Red Kite not attempting to breed in any given breeding season). Secondly, recoveries (ringed birds subsequently found dead), and re-sightings of birds ringed under license over the period 1994–present as part of the BTO Ringing Scheme, many of which have also been colour-marked with wing tags and/or leg rings (Cross 2023), can provide an understanding of typical intra- and inter-annual movements made by individual Red Kites.

Aims

In this study, we employ a PVA approach to model and predict the impacts of concept-phase and formally-proposed wind farm developments on the Welsh Red Kite population. We present results in comparison to a baseline scenario, which effectively incorporates existing wind turbines in Wales while assuming no further wind turbines are commissioned, at a range of spatial scales to aid interpretation of impacts upon national, regional and SPA-level populations. We attempt to identify the level of additional mortality that, all else being equal, would be required to halt population growth in the Welsh population of Red Kite.

METHODS

Impact scenarios and outcome scales

Throughout this study, we aimed to test the impacts of a range of wind farm development scenarios, and to describe these across a range of informative spatial scales. Our input scenarios were: (i) the most extreme case, in which every development currently in concept-phase or formally-proposed (across every developer involved in the study) is implemented, and (ii) the total impact of each individual developer. We described the outcome of each of these scenarios: (i) nationally, (ii) for each NRW Area Statement area (a subdivision of Wales into six terrestrial zones, allowing regionally-important environmental issues, priorities and opportunities to be identified and incorporated into NRW's work at the national scale), and (iii) for the population associated with the SPAs designated for Red Kites. Because the SPA populations of Red Kites are often considered to include all individuals within a 2km zone surrounding the SPA, we tested impacts on SPA populations both a) solely within the SPA boundary, and b) within the SPA boundary plus a 2km buffer.

We additionally aimed to identify the national-scale rate of additional mortality from wind turbine collisions that would result in the cessation of the current growth of the Welsh Red Kite population (i.e. population stabilisation of the median population trend estimate).

Data requirements for Population Viability Analysis

All analyses were conducted in R version 4.2.2 (R Core Team 2022). Population Viability Analysis (PVA) models require detailed demographic data, which was collated from BTO data, published literature, Welsh Kite Trust reports and data held by individual ringers. Final data sources are listed and referenced below, and for each data source, any initial selection and/or filtering process applied before analytical selection is briefly summarised. Within-study analyses included dispersal ranges split between two age classes and lifetime skip rate for breeding attempts.

Population and distribution estimates

Distribution estimates for Red Kite, in the form of interpolated population density, were sourced from the 2007-11 BTO BirdAtlas (Balmer *et al.* 2013). Estimates are at “tetrad” resolution (i.e. 2 x 2km grid squares) and represent the relative density of individuals during the breeding and winter seasons; for this study, we used the winter estimates since individual Red Kites are expected to remain in the vicinity of nest sites during the breeding season (acting as central-place foragers), but range more widely during the winter, thus potentially encountering more wind farms. Fieldwork for the BirdAtlas took place across the entire period 2007-11, so estimates are nominally centred on the year 2009. Given the time elapsed since this survey period, and the status of Red Kite as a re-colonising species within Wales, we cross-referenced these distribution estimates against up-to-date observation data for Red Kite, supplied by Cofnod and collated by Local Environmental Records Centres Wales (<https://www.lercwales.org.uk>). Specifically, we overlaid BirdAtlas distribution maps with maps of observations, and by visual inspection, ensured that there were no regions which had few Red Kites during the Atlas survey period, but had many subsequent observations.

National-scale population estimates for Red Kite were sourced from the Avian Population Estimates Panel’s fourth report (APEP4; Woodward *et al.* 2020), referring to populations in the year 2016. Estimates for Great Britain only (rather than for the entire UK) were used. Contemporary population estimates were produced by adjusting the APEP4 population estimates according to population trends for Red Kite in the intervening years, using annual population indices for Wales from the Breeding Bird Survey (Heywood *et al.* 2023).

Broadly, population estimates from APEP4 (provided in numbers of breeding pairs, and doubled where necessary to represent numbers of breeding individuals) were used to calibrate the interpolated population density surface derived from BirdAtlas, such that a population estimate could be derived for each GB tetrad. These estimates were extrapolated using the annual BBS index for Red Kite in Wales to provide a population estimate for 2022. This approach assumes that local population trends – for example, those at tetrad-scale or within any other spatial boundary – reflect national population trends.

Population estimates for each individual tetrad could then be summed to provide baseline population estimates for different spatial boundaries (Table 1). These included: (1) a national estimate for Wales, and (2) estimates for each NRW Statement Area, (3) for each of the two SPAs with Red Kite listed in their designations, and (4) for the area potentially impacted by each individual proposed wind farm development (which was the footprint of the site itself, plus a buffer zone related to the dispersal distances of juvenile and adult Red Kites; see below). The results of these individual development-specific analyses are not presented in this paper, but were confidentially communicated to each developer.

Category	Spatial boundary	Population estimate Pairs
Wales	Wales national boundary	2117
	Berwyn SPA boundary	4
Wales SPA	Berwyn SPA plus 2 km buffer	10
	Elenydd-Mallaen SPA boundary	79
	Elenydd-Mallaen SPA plus 2 km buffer	179
	North West Wales	99
NRW Area Statement Areas	North East Wales	3
	Mid Wales	1488
	South West Wales	468
	South Central Wales	22
	South East Wales	9

Table 1. Baseline (2022) Red Kite population estimates for relevant spatial areas, estimated using data from BirdAtlas, APEP4 and BBS, and used as a starting point in PVA modelling.

This approach generated a population estimate for Red Kite in Wales in 2022 of 2,117 breeding pairs, which is not dissimilar from the 2,500 pairs estimated by the Welsh Kite Trust in 2019 (Jenks 2020). Population estimates for the two SPAs were larger than the maximum number of nests recorded in these areas during the 14 years of intense nest recording (2000–2013) undertaken by the Welsh Kite Trust (Elenydd-Mallaen SPA: 18 nests in 2011; Berwyn SPA: zero nests in all years monitored), which is to be expected since Red Kite nests can be hard to detect, and observer effort is constrained by available resources, so there are likely to be more pairs present than nests recorded. For both SPAs, we estimated that there were at least as many Red Kites present within a 2km buffer surrounding the SPA than in the entire SPA itself (Table 1). This was also the case among nests recorded by the Welsh Kite Trust, with a peak of 88 nests recorded in 2011 in the Elenydd-Mallaen SPA plus a 2km buffer (of which 70 nests were in the buffer zone), and a peak of 4 nests recorded in 2006 in the Berwyn SPA plus a 2km buffer (of which all 4 nests were in the buffer zone).

Demographic rates from the published literature

Most demographic parameters for Red Kite were sourced from the published literature. We required estimates of the range of plausible values for the population mean in Wales for each of several parameters: maximum brood size, age at first breeding, (annual) productivity, breeding skip rate, and annual survival rates of four age classes (adults, 1st, 2nd and 3rd year immatures respectively). We collated published estimates of all variables and determined plausible ranges for each parameter (Table 2, p.82).

Breeding skip rate

Data from the Welsh Kite Trust’s long-term nest monitoring dataset (Welsh Kite Trust 2023) was used to estimate breeding skip rates in Red Kites in Wales. Skip rate is the proportion of breeding individuals that do not make a breeding attempt in any given year; this variable is required by PVA models. The data consisted of annual recording at known territories, with survey effort recorded such that years in which surveys took place, but no active nest could be identified, are distinguishable from years in which surveys did not take place. We first filtered the dataset to identify territories which had been continuously monitored for at least four years. Since the first and last year of recording at any given territory could not be confidently labelled as a ‘skip’, we then restricted the data further, to a 10-year period between 2002-11 that was internal to the overall data range. For each year in this period, we calculated the proportion of established territories that had been monitored, but where no active nest was identified. We used these annual estimates of skip rate to determine a plausible range for PVA modelling (Table 2).

Parameter	Plausible range	Source
Maximum brood size	2-3	Pfeiffer and Mayberg (2015) Davis and Newton (1981) Jørgensen (1989) Evans <i>et al.</i> (1999)
Age at first breeding	2-3	Davis and Newton (1981) Evans <i>et al.</i> (1999) Smart <i>et al.</i> (2010) Tenan <i>et al.</i> (2012) Bellabaum <i>et al.</i> (2013) Ceccolini <i>et al.</i> (2013) Mammen <i>et al.</i> (2014)
Productivity	1-2	Annual reports of the Welsh Kite Trust (e.g. Welsh Kite Trust, 2017)
Adult annual survival	0.75-0.95	Newton <i>et al.</i> (1989) Evans <i>et al.</i> (1999) Desholm (2006) Schaub (2012) Katzenberger <i>et al.</i> (2019) Pfeiffer and Schaub (2023)
Immature survival 1st year	0.516-0.804	Carter (2007) Evans <i>et al.</i> (1999) Carter and Grice (2000)
Immature survival 2nd year	0.65-0.944	Duffy and Urquhart (2014) Evans <i>et al.</i> (1999) Carter and Grice (2000)
Immature survival 3rd year	0.73-0.927	Evans <i>et al.</i> (1999) Carter and Grice (2000)
Breeding deferral/skip rate	0.085-0.185	Calculated in this study from territory monitoring data (Welsh Kite Trust, 2023)

Table 2. Demographic parameters used in Population Viability Analysis models. Parameters derived in the present study, rather than sourced from the literature, are in bold italics.

Spatial ranges

Data from a long-running ringing and colour-marking project (Cross 2023) was used to estimate dispersal distances in Red Kites in Wales. This dataset consisted of re-sightings of live birds and “recoveries” of dead birds, for 1,082 individual Red Kites ringed under license in Wales as part of the BTO Ringing Scheme (Robinson *et al.* 2023). Dispersal distance, for the purposes of this project, was defined as the typical maximum distance travelled by either (a) a juvenile Red Kite from its natal nest, or (b) an adult Red Kite from its nest site. For this project, dispersal distances were required to determine the spatial area across which each wind farm proposal could impact upon resident Red Kites. To this end, we calculated the distances between each resighting/recovery for a bird and (a) its original ringing location, for re-sightings of juveniles (i.e. less than one year after ringing as an unfledged pullus in the nest), or (b) each other resighting/recovery after one year of age, for adults. For each age group, we filtered out re-sightings that were less than 1km apart (and therefore may have been associated with repeated sightings at the same location), and then calculated the 75th percentile of between-points distances; we considered this to strike a balance between capturing the typical distances between locations that might realistically be travelled by a majority of individuals, yet excluding the extreme distances travelled by a very small subset of individuals. We repeated this process twice, first excluding re-sightings at Red Kite feeding stations, and second including all re-sightings for birds that had been seen at least once at a feeding station. This enabled us to estimate the dispersal distance of Red Kites in a typical landscape, but also to understand whether birds may fly further in order to access feeding

stations (with implications for proposed developments in the vicinity of such feeding stations).

Spatial boundary data

We obtained a range of spatial boundaries (in GIS formats) to facilitate analysis and reporting at different scales. Spatial boundaries for the two SPAs designated for Red Kites, and for the NRW Area Statement regions, were downloaded directly from Datamap Wales (<https://datamap.gov.wales>).

Each wind farm developer involved in the project was asked to provide (under a Non-Disclosure Agreement) the locations of their current concept-phase and formally-proposed projects (preferably as a GIS shapefile of each proposed site’s boundary), along with metadata including the number of turbines proposed for each site, and the outcome of a Collision Risk Model (CRM) if one had been undertaken to date. Some developers also supplied this information for already-operational sites to support the imputation of collision risk at proposed sites where CRM had not yet been undertaken (see below).

We analysed data from all developers in combination, and also on a developer-by-developer basis. Summary metrics for each developer are reported non-identifiably.

Impacts on survival using Collision Risk Model data

To predict the impact of wind farm developments upon Red Kite populations, it was necessary to understand the rate of additional expected mortality (above and beyond natural mortality rates) associated with each proposed development in the study. Some developments had already undertaken Collision Risk Modelling (CRM), a standard model typically based on site-specific field observation data (Band *et al.* 2007; e.g. Urquhart and Whitfield 2016). However, many proposed developments had not yet reached this stage. Additionally, the outputs of standard CRMs are expressed as absolute numbers of birds expected to collide with turbines per year, whereas PVA requires this to be expressed as a risk of collision per exposed bird.

To translate absolute collision risk (CR) into a risk of collision per exposed bird, we divided the CR value for each site at which an estimate was available (15 sites in total, which included several already-operational sites that were otherwise outside the scope of this study) by the estimated baseline population for the site’s impacted area. This yielded a CR value per exposed bird that was directly equivalent to a rate of additional mortality expected as a result of exposure to the development.

To estimate the risk of collision for developments where no CRM had yet been undertaken, we fitted a simple linear model with the risk of collision per exposed bird predicted by the number of turbines in the development (excluding two sites that were clearly outliers in terms of their unusually high CR value per exposed bird). We found there was no significant relationship between development size (expressed as number of turbines) and the per-bird additional mortality rate (Figure 1, p.84), even though larger developments tend to have higher absolute CR values, probably because larger developments also had a higher number of exposed birds (i.e. absolute CR values were correlated with number of exposed birds). Therefore, we deemed it appropriate to apply an average value for the CR per exposed bird of 0.0027 to all developments, regardless of the number of turbines (i.e. roughly one bird per 400 exposed, on average, is expected to collide with a development each year, where larger developments tend to have a greater number of birds exposed).

In some cases, there was spatial overlap between the impacted areas (based on Red Kite dispersal distances) of multiple proposed developments. To allow us to model the population impacts of multiple developments, each with its own unique impacted area, on regional populations, we calculated a collision-risk estimate for each individual tetrad in Wales under each modelled scenario. For each tetrad,

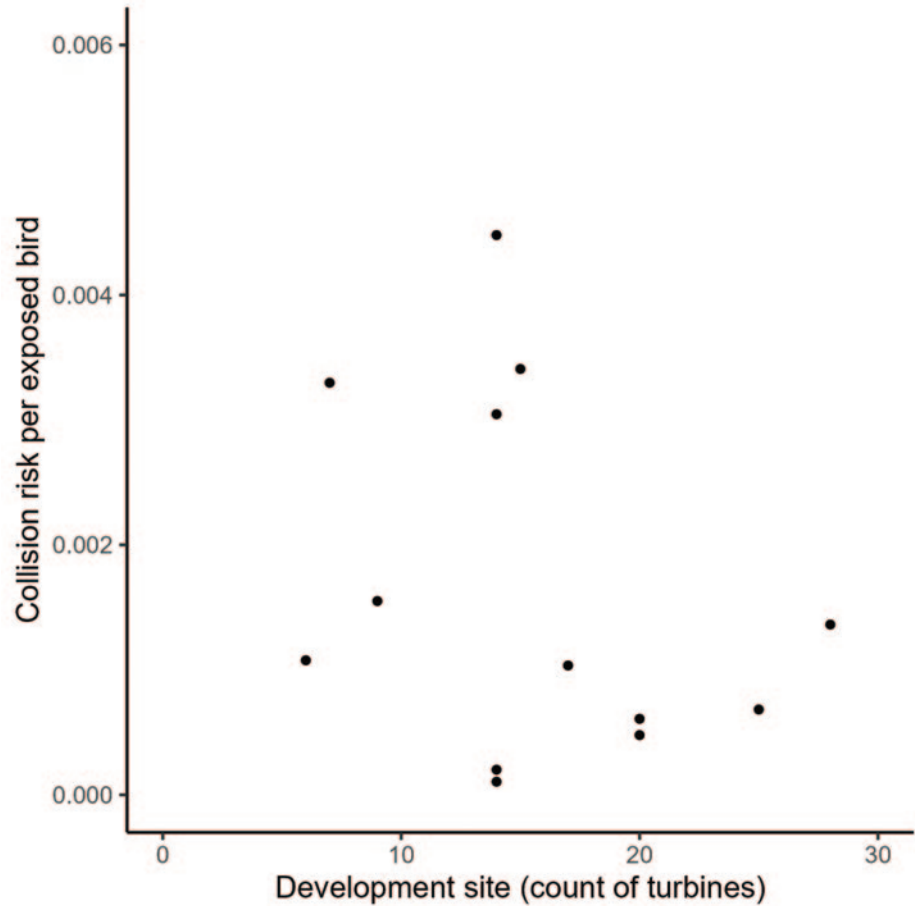


Figure 1. No significant relationship between development size (no. turbines) and CRM value per exposed bird. Larger developments tend to have higher CRM values as a function of having a greater number of birds exposed to risk. N.B. two outliers with unusually high collision risks were removed.

this collision-risk estimate was the multiplicative combination of collision risk for all developments expected to impact upon that tetrad; for example, a tetrad within the impacted area of two developments would have a collision-risk estimate of $1 - (1 - 0.0027)^2 = 0.0054$ (i.e. 0.54% of that tetrad’s population are expected to collide with a wind turbine per year).

Population Viability Analysis

Overview

Population Viability Analysis was conducted using the NEPVA R package (Searle *et al.* 2019), version 4.17, following the multi-model approach of Macgregor *et al.* (2022). In brief, we fitted a large number of deterministic models (using different combinations of plausible values for the various demographic parameters), and used a rejection sampling procedure to filter these models according to how well

their predicted population trend matched the recent Red Kite population trend in Wales, to generate a set of accepted baseline models. We then applied a range of impact scenarios to each baseline model to estimate the full range of possible outcomes arising from different starting population trends affected by different numbers of wind farm developments. Finally, we combined the most relevant set of population trends with starting populations at tetrad-level, in order to make population predictions that incorporated spatially-explicit variation in both Red Kite starting populations and impacts of proposed wind turbine developments.

Modelling environment

A range of options are available within the NEPVA package to conduct PVAs in different ways. Following Macgregor *et al.* (2022), the following options were employed in this study. Environmental and demographic stochasticity were eliminated from models (by setting `model.envstoch = “deterministic”` and `model.demostoch = FALSE`), because it was considered that too little data was available to make accurate estimates of intra- and interannual variability in productivity and survival. Instead, estimation of model uncertainty was conducted by modelling large numbers of combinations of input parameter levels and inspecting variation between outcomes (see below). Models were constructed without density dependence (`model.dd = “nodd”`) because Red Kite is still recolonising in Wales, and with productivity constrained by brood size (`model.prodmax = TRUE`). It was deemed not possible to model the potential onset of density dependence as the population approaches its carrying capacity, due to a lack of evidence to suggest what the carrying capacity for Red Kite in Wales is likely to be. The option to include a number of “burn-in” years to allow age structure within the model to settle was used, because the existing age structure of the Red Kite population is unknown; a value of 5 burn-in years was used because preliminary model runs showed this was sufficient to allow age structure in the models to stabilise. Because each model was fully deterministic, there was no need for multiple simulations of each model (`sim.n = 1` and `sim.seed = NULL`). Like Macgregor *et al.* (2022), we treated breeding skip rate as a modifier to per-nest productivity since the skip rate argument “`demobase.bskippc`” did not appear to be functioning as intended. Therefore, for each model, productivity rate was multiplied by $(1 - \text{skip rate})$ to determine the modelled productivity rate.

Models were fitted with a starting population size of 2,117 (breeding pairs), based on the estimated Welsh population size, but this was effectively arbitrary because tetrad-specific starting population sizes would be applied at a later stage in analysis. Impacts of collisions with new wind turbine developments were modelled to begin in 2025 and continue annually. The population trend of each model was calculated using a generalised linear model (Poisson error distribution with log link function).

Model construction and rejection sampling

The above procedure generated a total of 37,500 possible combinations of parameters to be modelled. Each model parameter provided five input levels – upper bound, lower bound, median, upper and lower quartile values, based on the plausible ranges in Table 2. We modelled all possible combinations of these values for all parameters except maximum brood size and age at first breeding, for which we modelled only the lower and upper bounds (since non-integers are nonsensical for these parameters).

Wind turbine mortality associated with new/future developments was applied as a modifier to survival of all age classes. In reality, it is possible that juveniles have a higher risk of colliding, since they may, for example, range further than territory-holding adults, and be still developing their flight capabilities, but there is currently no data on age-specific collision risks in Red Kite with which to assess this. For every baseline model, we applied a broad range of survival modifiers as follows (all values represent an additional percentage of the population killed by collisions with new wind turbines per year): increments of 0.025 between 0 and 0.1; increments of 0.25 between 0 and 1; increments of 0.5 between 1 and 9; increments of 1 between 10 and 20; and an additional 30 % scenario. These modifiers

were intended to encompass the full range of tetrad-level collision risk estimates (calculated as above) under all development scenarios, with the higher-mortality modifiers ensuring that the rate of additional mortality that would result in stabilisation (i.e. population growth halting) of the Welsh population of Red Kite could be identified. Note that collisions with existing, already-operational wind turbines are considered to be encompassed within the ‘baseline’ case where no additional survival modifier was applied.

Some combinations of parameters (e.g. if the upper bounds were used for all three of productivity, adult and immature survival) produced population estimates in the baseline (0% additional mortality) case that were clearly not consistent with observed population trends for Red Kite in Wales (e.g. population explosion or decline). Although each parameter within these models was individually plausible, in such cases the overall combination was not plausible. To narrow down the total set of 37,500 candidate models to a set of models that could plausibly explain observed population trends, we calibrated baseline models against population trends from the BBS, using Approximate Bayesian Computation (Hartig *et al.* 2011). We used ten-year trends calculated over the period 2012–2022 for all BBS squares in Wales where Red Kite was observed, and applied a rejection sampling approach whereby the probability of each model’s acceptance was determined by the Probability Distribution function (PDF) of the 10-year trend estimates, which was approximated by bootstrapping (Heywood *et al.* 2023). See Macgregor *et al.* (2022), Figure 1, for a visual representation of this process. This means that models predicting a rate of population change similar to the central BBS trend estimate (67 % increase over 10 years; 95 % confidence interval 32–112 %) were relatively more likely to be accepted, and models predicting a rate of change far from the central BBS trend estimate (i.e. the extremes of the PDF) were more likely to be rejected.

Spatially-explicit population predictions

To translate population trend projections from our set of accepted generic PVA models into population trend projections that incorporate spatially-explicit variation in both starting populations and wind turbine impacts, we first extrapolated the population of each individual tetrad under each scenario. Selecting a model at random from the accepted set (see below), we selected the survival modifier that was the closest match to each tetrad’s individual collision risk estimate and used the relevant modelled population trend to project that tetrad’s starting population forward in time over 10 years. We summed the annual tetrad populations for all tetrads in each outcome scale (e.g. for the national-scale outcomes, we summed annual population estimates for all tetrads in Wales to generate a prediction of the Red Kite population in each year for 10 years). We then fitted a generalised linear model (Poisson error distribution with log link function) to these summed annual population estimates to identify the overall population trend within each outcome scale. We applied this procedure to all accepted PVA models in order to describe 95% confidence intervals for population projections under each impact scenario and at each outcome scale.

Summarising outcomes

To aid with interpretation, we summarised outcomes in several ways. First, we depicted the 95 % confidence intervals for population projections under each impact scenario, at each outcome scale, as population trend lines on graphs in direct comparison to the baseline (unimpacted) scenario.

In addition to describing the 95% confidence intervals of projected trends as above, we categorised models according to which of five outcomes they resulted in, over a 5-year timescale: decline (< 0% increase), stability (0–25% increase), low growth (25–50% increase), medium growth (50–100% increase), and high growth (> 100% increase). We then calculated the proportion of all models under each impact scenario, at each outcome scale, which resulted in each category, presenting these in direct comparison to the baseline (unimpacted) scenario in each case.

Lastly, we projected the 95 % confidence interval for the size of the Welsh population of Red Kite by the year 2050, under each scenario and at each outcome scale, presenting scenarios in direct comparison to each other, as well as in comparison to the baseline (unimpacted) scenario.

RESULTS

Breeding skip rate

The annual average breeding skip rate based on site occupancy varied between 8.6% (2004) and 18.2% (2003), with no apparent trend over time (Table 3). This is broadly comparable to the 9 % skip rate estimated by Pfeiffer and Schaub (2023). Based on this, a skip rate range of 0.085–0.185 was used in PVA modelling, to encapsulate annual variation in the recorded range. This is likely to represent an overestimate of the real skip rate to some extent, because Red Kite nests can be hard to detect even within known territories and some that failed early, prior to monitoring visits, may have been misinterpreted as skips.

Year	Annual breeding site skip-rate	
	Proportion	Percentage
2002	0.13	13.33
2003	0.18	18.18
2004	0.09	8.57
2005	0.18	18.06
2006	0.13	12.70
2007	0.16	16.00
2008	0.14	14.12
2009	0.10	10.13
2010	0.16	16.25
2011	0.14	14.47

Table 3. Annual variation in skip rates calculated for Red Kite territories with long-term annual monitoring in Wales.

Spatial ranges

On average, we found that immature Red Kites ranged further from the nest than adults. There was little difference between (i) the distances ranged by all immature Red Kites, with re-sightings at feeding stations excluded from the dataset, and (ii) those ranged by the subset of immatures that visited feeding stations, with all re-sightings included. However, the distances ranged by all adults, with re-sightings at feeding stations excluded from the dataset, were half those ranged by the subset of adults that visited feeding stations, with all re-sightings included (Table 4, p.88).

The 75th percentile of between-resighting distances for immatures and adults respectively were 21.7 km and 16.3km (in other words, only 25% of all immature Red Kites range further than 21.7km from their natal nest during their first year, and only 25% of all adult Red Kites range further than 16.3km from their nest site). Based on these distances, we conservatively assigned a buffer distance of 22km around all proposed development sites in order to determine the potentially impacted area. Whilst this is a relatively large buffer given the outcomes of this analysis, it represents a cautious approach through which we capture the vast majority of plausible mortality associated with each wind farm development.

Age category	Distance between the nest and re-sightings (km)			
	n observations	25 th percentile	50 th percentile (median)	75 th percentile
Immature (exc. feeding stations)	499	4.3	8.3	21.7
Immature (using feeding stations)	2275	2.6	5.6	22.7
Adult (exc. feeding stations)	585	2.1	4.2	16.3
Adult (using feeding stations)	657	7.1	13.7	33

Table 4. Average distances of individually-marked Red Kites originally marked in Wales for immatures (between the natal nest and re-sightings during their first year) and adults (between re-sightings after one year of age). These are split between all birds with visits to feeding stations excluded, and for all re-sightings of birds that visited feeding stations.

Population Viability Analysis

In total, the rejection sampling procedure selected 19,775 plausible accepted models for Red Kite (53% of the 37,500 candidate models). The vast majority of baseline models predicted population stability or increase.

Impacts of wind farm development on Red Kite population trends

Modelled impacts on the mean Red Kite population trend from all the sites provided by developers are presented in Figures 2–4. Across virtually all combinations of impact scenarios and outcome scales, impacts are projected to be relatively small, even under the scenario where all developers’ plans were modelled in combination (Figure 2). Every impact scenario is predicted to be less likely to result in population decline than continued growth at all outcome scales. In general, modelled impacts were larger for spatially-smaller outcome scales with relatively high-density Red Kite populations (e.g. the two SPA populations, in both cases taken to include birds within a 2km buffer zone around the SPA boundary). Impacts were larger for the higher-density Elenydd-Mallaen SPA population than for the Berwyn SPA population, where the current density of Red Kites is lower.

Impacts for the all-developers scenario were substantially larger than for any one developer taken in isolation, but there was also significant variation between the projected impacts of different developers’ plans. Developers projected to have the largest impacts were those with large numbers of developments in proposal (or spatially large developments), and as above, where those developments were located in areas with high densities of Red Kites (e.g. in the vicinity of the Elenydd-Mallaen SPA).

These patterns were also evident when considering the relative probabilities of different population outcomes under different impact scenarios compared to the unimpacted baseline (Figure 3, p.90). The implementation of all development sites in combination would increase the probability of the Welsh population of Red Kite stabilising from 17% to 28% (Figure 3a), and even introduce a possibility of population decline in each SPA population where none exists under the baseline scenario (0.4% probability of decline in the Berwyn SPA population; 4% probability of decline in the Elenydd-Mallaen SPA population; Figure 3b–e). Note, however, that in all cases there remains a greater than 50% probability of continued population growth. Again, each individual developer’s portfolio had a lesser impact than the combined effect of all developers, with substantial variation between developers.

Whilst impacts upon population trends may appear relatively small, these may propagate into substantial differences in population size given enough elapsed time (Figure 4, p.91). For example, the median population size estimate for the Elenydd-Mallaen SPA in the year 2050 under the all-developers scenario was approximately half of the equivalent estimate under the unimpacted baseline, even

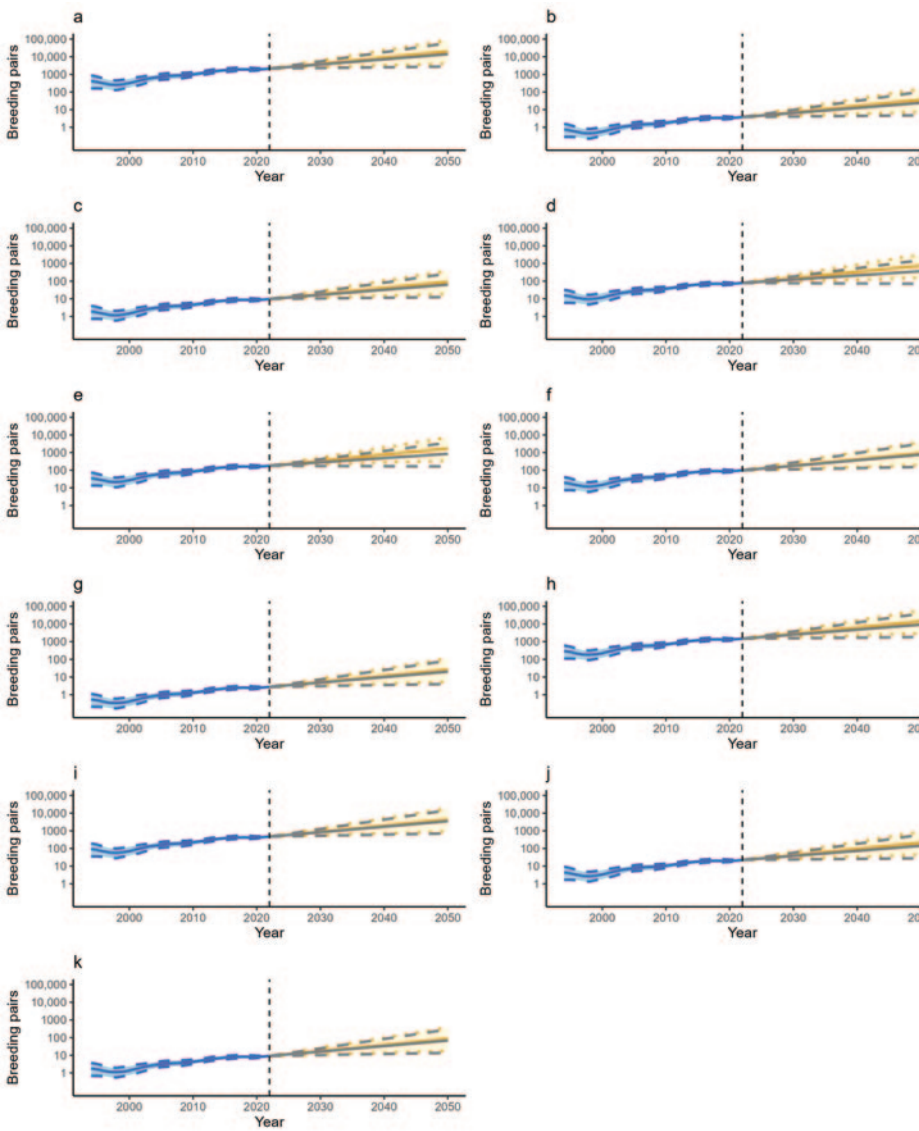


Figure 2. Projected impacts of all concept-phase and formally-proposed wind farm developments in combination at a range of outcome scales. The recent Red Kite population trend (from BBS trend for Wales) is shown in blue. Baseline (unimpacted) population projections are shown in grey, and the impacted population projections are shown in yellow. For all line colours, dotted/dashed lines and shading (blue and yellow only) show 95% confidence intervals. Panels show projections at different outcome scales: a) all Wales; b) Berwyn SPA; c) Berwyn SPA plus 2 km buffer; d) Elenydd-Mallaen SPA; e) Elenydd-Mallaen SPA plus 2km buffer; f) North West Area Statement (AS); g) North East AS; h) Mid-Wales AS; i) South West AS; j) South Central AS; and k) South East AS. Note that the y-axis is shown on a logarithmic scale.

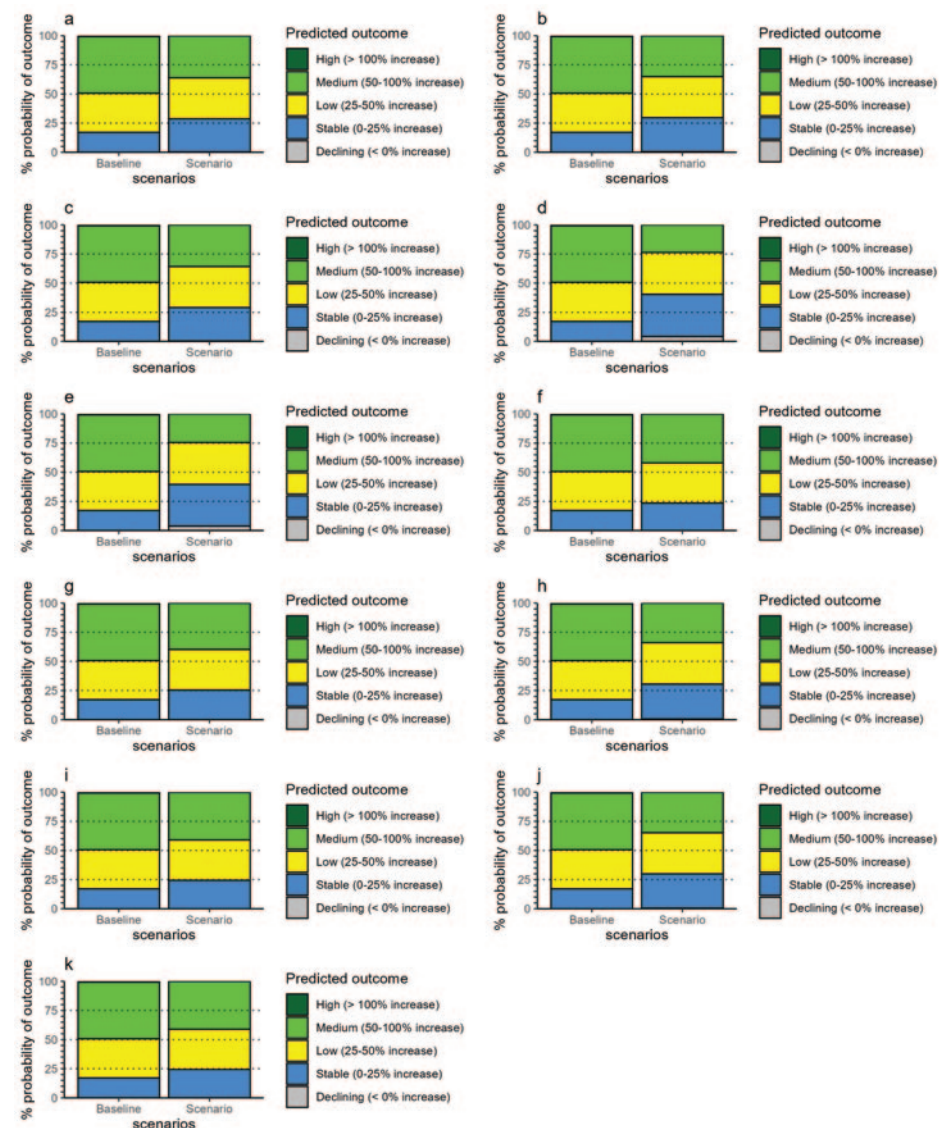


Figure 3. Relative probabilities of different population outcomes for all concept-phase and formally-proposed wind farm developments in combination at a range of outcome scales. In each panel, the impacted scenario (right) is compared to the unimpacted baseline (left). Outcomes are grouped into high, medium and low rates of population increase (shaded in dark green, light green and yellow respectively); population stability (shaded blue) and population decline (shaded grey). Panels show probabilities at different outcome scales: a) all Wales; b) Berwyn SPA; c) Berwyn SPA plus 2 km buffer; d) Elenydd-Mallaen SPA; e) Elenydd-Mallaen SPA plus 2 km buffer; f) North West Area Statement (AS); g) North East AS; h) Mid-Wales AS; i) South West AS; j) South Central AS; and k) South East AS.

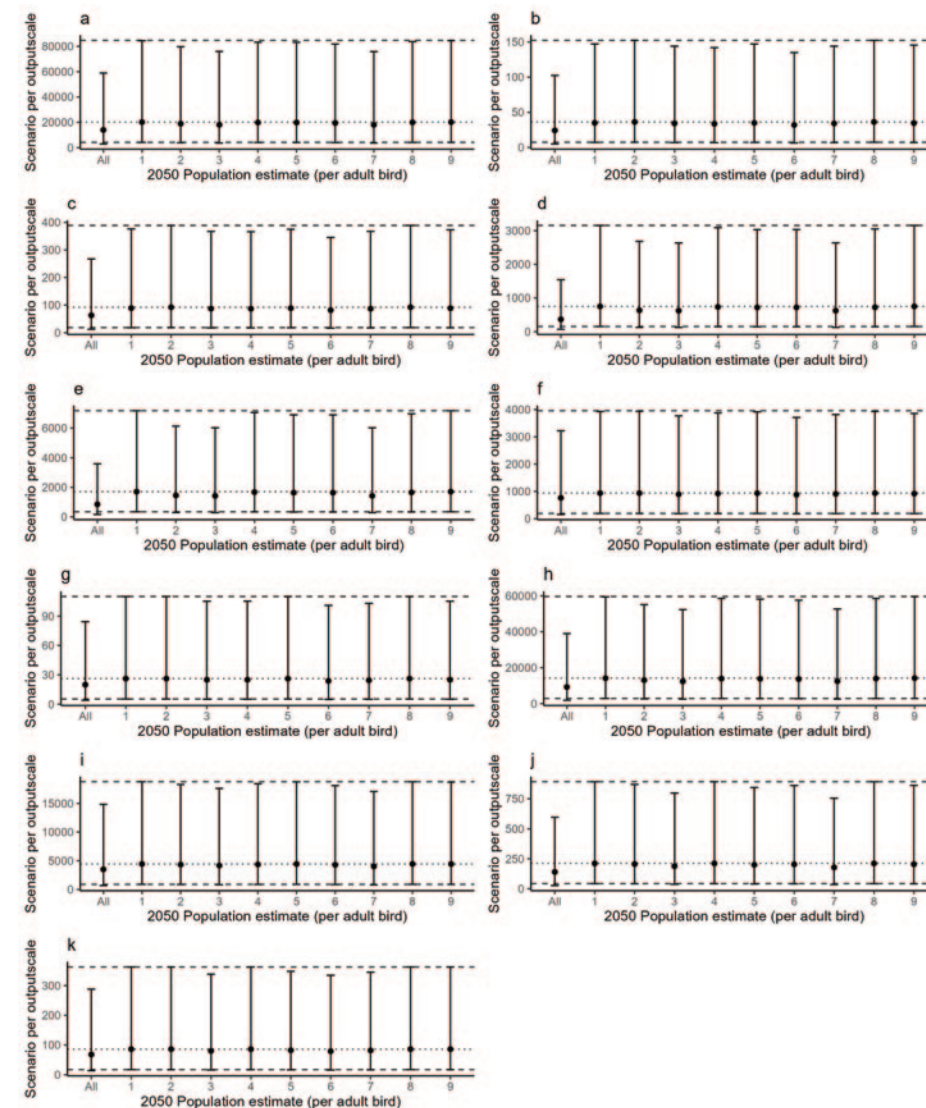


Figure 4. Projected Red Kite populations in the year 2050 under each impact scenario, at each outcome scale, with no onset of density dependence. Horizontal lines show projected populations under the unimpacted baseline scenario (dotted: median, dashed: 95% confidence interval). Points show median estimates and lines indicate 95% confidence intervals for each impact scenario. Panels show projected population sizes at different outcome scales: a) all Wales; b) Berwyn SPA; c) Berwyn SPA plus 2km buffer; d) Elenydd-Mallaen SPA; e) Elenydd-Mallaen SPA plus 2km buffer; f) North West Area Statement (AS); g) North East AS; h) Mid-Wales AS; i) South West AS; j) South Central AS; and k) South East AS. In each panel, a population size is shown for the 'all developers' impact scenario, as well as for each developer in turn (labelled non-identifiably as developer 1–9) along the x-axis.

though both estimates were greater than the current (2022) population estimate, and therefore represent population growth. Note that our modelled scenarios effectively assume that the Red Kite population does not reach carrying capacity by 2050 (and therefore there is no onset of density dependence).

Red Kite population tolerance to decreased adult survival

None of the scenario-focussed analyses resulted in a prediction whereby population decline was more likely than increase. Analysis of more extreme survival modifiers suggests that an additional 12–13% mortality per year would be necessary to lead to the median population model projecting a decline (Figure 5). This is far in excess of the levels associated with current proposed developments, even at very local scales, let alone as a national average.

DISCUSSION

Implications of modelled results

The key finding of this study is that Red Kite population stabilisation or decline in Wales, solely as a consequence of wind energy development, is unlikely. Even under the most extreme scenario tested, where every project currently in development by all nine collaborating developers (to our knowledge, representing most of the developers active in mid-Wales) is ultimately approved and commissioned, our modelling suggests that there is approximately a 28% probability of population stabilisation, and virtually no chance of decline (Fig. 3a). An average of 12% of the national population could be killed

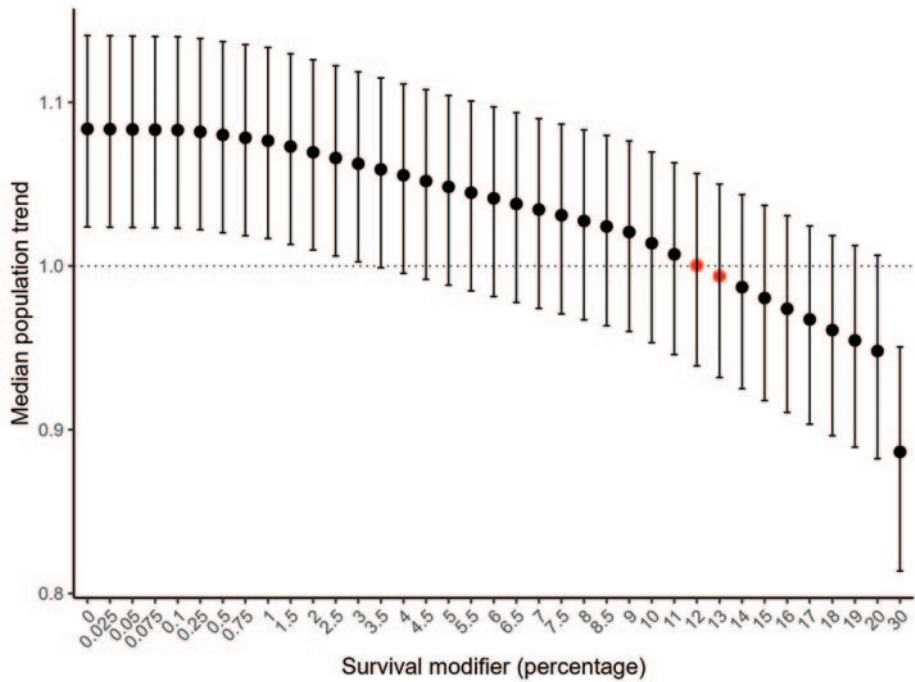


Figure 5. Population trends projected using each survival modifier modelled using PVA. Points show the trend of the median model, with lines indicating the 95% confidence interval. Red points highlight the values between which the median model projections transition below 1 (dotted line), where a population trend of 1 indicates population stability. Note that the x-axis is categorical rather than continuous.

per year in collisions with new wind turbines before our models would suggest population decline as being more probable than not, all else being equal (Figure 5).

Nevertheless, our models suggest that more caution is warranted with respect to developments in close proximity to the Berwyn and Elenydd-Mallaen SPAs. The impacts of additional mortality from high densities of wind turbines adjacent to the relatively small, high-density populations of these protected sites were distinctly greater. In particular, decline of the Elenydd-Mallaen SPA population was assessed to have a probability of approximately 4% under the all-developers scenario (Figure 3e), with a lower probability of 0.4 % for decline of the Berwyn SPA population (Figure 3c); these were the only analyses in the entire study to produce a probability of population decline that was greater than zero. Each individual developer scenario forecast a zero probability of decline, emphasising the importance of evaluating impacts of multiple developments in combination. This caution extends some distance beyond the boundary of each SPA, given that at least as many Red Kites are present within the 2km buffer zone around the SPAs as within the designated areas themselves, for both the Berwyn and Elenydd-Mallaen SPAs (Table 1). Predicted impacts were in all cases comparable when considering Red Kites within the SPA boundary itself, or when considering the ‘SPA population’ including a 2km buffer zone around each SPA.

Limitations to predicting impacts using an extrapolated population trend

Predicting future population trends from historic survey data (in this case, with rejection sampling of candidate models calibrated using trends from the Breeding Bird Survey) assumes that conditions in the future apply exactly the same positive and negative pressures on a species as have been applied in the past. Although most ecological pressures will act at local scales, and therefore have only weak effects on national populations, it is possible that new and unforeseen pressures may act on the Red Kite population and weaken model performance against the surveyed population trend in future. In order to illustrate this issue, we present three different potential scenarios likely to modify the population trend in future, that cannot, at present, be incorporated into the demographic modelling exercise.

First, the Welsh Red Kite population is still recovering from historic catastrophic decline to near-extinction, and has not yet reached its carrying capacity (the maximum population that can be supported by contemporary landscapes). As the population approaches this ecological limit in future, density-dependent effects may come into force, such that some demographic parameters will change as they become limited by available resources. For example, if the limiting resource is food availability in winter, we might see a decline in immature survival, perhaps followed by a decline in adult survival. Under such a scenario of density-dependent effects on survival, it is possible that some level of mortality associated with wind turbine collisions could be sustainably offset by increased survival among the remaining birds. If food resources during the breeding season become limiting, we might see decreased nest productivity, chick survival or clutch sizes. If nest-site availability decreases and/or becomes limiting, we might see increases in age at first breeding, or larger spatial movements from immature birds because all available territories around their natal sites are occupied. The most likely scenario is that all of these parameters would change to a greater or lesser extent as the population continues to grow; an adjustment that is impossible to predict from current data. There is also likely to be considerable regional variation in the timing of when different stages of this recovery-adjustment process are reached. At the centre of their range, local populations may already be stable or nearly so, while at the range margins in North and parts of South Wales, where Red Kites are still largely absent or only present at low densities, the species is still expanding into landscapes where resources are less limited. One of the benefits of a multi-model approach is that there are individual models within the thousands of plausible models for the population that reflect all these possibilities, from the core (stable) sub-populations to the rapidly increasing ones.

Another likely future includes new pressures associated with large-scale processes such as climate change, novel tree diseases, and land-use change. We are already seeing changes in weather patterns, seasonality and phenology, and in some species there are clear links between, for example, stormy weather in the breeding season and years of poor productivity (*cf.* high rates of nest failure in Osprey *Pandion haliaetus* attributable to storm events; Forsys *et al.* 2021), novel tree diseases and changes in colonial behaviour (for example, the impact of Dutch elm disease on spatial distribution of rookeries; Brenchley 1986), and of course agriculture and farmland bird population changes (Burns *et al.* 2020). The impacts of these pressures on Red Kite are difficult to predict in detail – we might hypothesize that a trend of increasing winter rainfall might reduce the survival of immature Red Kites, but not be able to predict either the climatic trend or its impact on Red Kites accurately enough to include it in a demographic model. An additional possible scenario that could lead to food limitation is the removal or closure of feeding stations; this would also likely result in changes to Red Kite dispersal and ranging behaviour, given that adults appear to travel further than normal in order to access feeding stations (Table 4).

A final pressure worth considering, although impossible to anticipate in detail, is the effect of a specific novel disease outbreak in Red Kite. Scavenging and predatory species are, for example, notably vulnerable to transmissible diseases, such as the recent multi-species epizootic of highly pathogenic avian influenza (HPAI). HPAI has had major impacts in the UK on seabirds, particularly colonial species (terns), scavenging species (Great Skua *Stercorarius skua* and some of the large gull *Larus* species) and large raptors, particularly Golden Eagle *Aquila chrysaethos* and White-tailed Eagle *Haliaeetus albicilla* (Falchieri *et al.* 2022, Wilson *et al.* 2023). There is evidence of HPAI incidence in Scottish populations of Red Kite (Wilson *et al.* 2023), but as yet, no data with which to explicitly parameterise the impacts of a disease outbreak in the demographic model. Nonetheless, our models suggest that any driver causing an additional 12% mortality in Red Kites could lead to decline in the Welsh population; although the context of this study is wind turbine collisions, this additional mortality could, in principle, come from any source or combination of sources. This percentage is directly comparable to, for example, the percentage of Great Skua known to have died during the recent HPAI epizootic (Falchieri *et al.* 2022). Therefore, the possibility exists that a significant disease outbreak could markedly change future population trends in this species, and this must be taken into consideration when interpreting the results of our study on the impacts of wind farms.

Conservative modelling of spatial movement data

Individual birds' movement patterns vary with a number of ecological and environmental factors, including age, sex, time of year, landscape structure, availability of (and dependence on) key resources such as feeding stations, and local population density. In general, individuals are most mobile during post-fledging dispersal when young birds are looking for territories of their own (Table 4). Individuals are also expected to be more mobile outside the breeding season (when breeding adults' movements are less constrained by nest location and landscape quality), but it is challenging to quantify such seasonal patterns using ring recovery data since there are often long time-intervals between consecutive re-sightings of a bird, obscuring any information about when (in the year) movements between locations have taken place. GPS-tracking datasets would therefore be better suited to understanding seasonal variation in Red Kite movements (e.g. van der Wal *et al.* 2015), but such data are not yet available for Red Kites in the Welsh landscape. Adult Red Kites also travel further (likely in all seasons) when they are able to take advantage of permanent feeding stations (Table 4), suggesting that greater caution may be justified when considering developments within approximately a 30km radius of a feeding station. Our modelling approach has a minimum time-step for population impacts of one year, so that mortality in immature or wintering birds is calculated as an impact on the following breeding season. It is therefore necessary to consider the distances travelled by individual birds during their most mobile life-stage and period of the year in order to adequately assess their risk of interacting

with wind farms.

It is likely that this approach somewhat over-estimates mortality risk in territorial adult birds during the breeding season, since there is some evidence that individuals learn to avoid turbines even when entering the site footprint of an established wind farm (Duffy and Urquhart 2014, Urquhart and Whitfield 2016). Since underestimating wind farm impacts carries more risk into decision-making than overestimating them, we elected to use the larger spatial scale of immature and winter distance movement to model impacted areas around development sites.

An additional caveat with use of PVA is the core assumption that the population being modelled is a closed population of breeding individuals. This assumption is likely violated for Red Kite in Wales, since there is known migration of some individuals to and from the English and other regional populations. Nevertheless, genetic studies (Skujina 2013) suggest that this immigration and emigration still represents a relatively small proportion of Welsh breeding birds, which are therefore somewhat reproductively isolated and distinct from the (reintroduced) English, Scottish and Irish populations. It does, however, introduce a note of caution, because of the possibility of asymmetric dispersal into and out of Wales leading to outcomes beyond those predicted by our PVA; for example, if immigration from the expanding English populations was sufficient to offset wind turbine mortality of Welsh birds. This emphasises the importance of understanding existing data on movement patterns (from existing ring and colour-mark recoveries), and potentially in the future using GPS tracking, to measure the rates of immigration and emigration between the adjacent Welsh and English Red Kite populations, and to use these to model the effects of dispersal in both directions on the robustness of the Welsh population to mortality caused by wind turbines.

Habitat-specific variation in collision risk

Collision risk modelling often predicts lower risk to Red Kites from turbines constructed in afforested locations. We were unable to incorporate this into our models since all sites for which CRM data were available to us were on open or mixed ground (none were purely afforested when overlaid with land cover data from Marston *et al.* 2022). Our analysis therefore effectively assumes that all future developments are on open ground, in a setting where they may have the most impact. As above, this is an appropriately conservative approach since underestimating impacts carries more risk into decision-making than overestimating them.

Breeding skip rate

Our analysis of breeding skip rate in the Welsh population of Red Kite relied on repeated monitoring of known territories over the period of several years in order to distinguish established pairs that skipped breeding for a year and then resumed from those that had permanently abandoned their territory or died. Whilst we believe this to be a robust approach, it is contingent on the assumption that surveyors for the Welsh Kite Trust were able to identify the majority of nesting attempts within known territories in any given year. Since Red Kites, despite being territorial, do not consistently return to the same precise nest location every year, and tend to build their nests high up in trees, it is quite likely that some recorded absences that we considered to be 'skips' for analytical purposes were in fact non-detections of pairs that did attempt breeding. This is particularly likely to be the case for nesting attempts that failed early in the breeding season (*i.e.* before the first monitoring visit). Therefore, our estimates of skip rate are likely to tend to be higher than the true value. Indeed, the one previous estimate of skip rate in Red Kite of which we are aware, 9 % (from Pfeiffer and Schaub 2023), falls near to the extremes of, but still within, the 8.5–18.5% range that we applied to our models. Since the rejection sampling approach calibrated our final model set against BBS trends, models may have been selected with higher skip rates than reality, compensated for by, for example, higher productivity or survival rates than reality. However, the change in survival rate that we modelled is the same, whichever survival rate underpins

a model, suggesting that the conclusions of our PVA are robust to this caveat.

Conclusions

The results of this Population Viability Analysis suggest that whilst collisions with wind turbines may be a cause of mortality in Red Kites, this mortality alone is unlikely to prevent the continued recovery of the Red Kite population in Wales. These results may provide some reassurance to policy makers, given significant concerns around the conservation implications of commissioning new onshore wind energy. Given that climate change is itself a significant conservation issue with the potential for difficult-to-predict, and in some cases severe, impacts upon a wide range of species, this issue represents a conservation trade-off between, on the one hand, the potential mitigating effects of increasing the renewable energy capacity of the National Grid, to reduce demand for fossil fuels, and on the other hand, the threat posed by wind turbine collisions to the Red Kite population. Our findings, that this risk is relatively low (albeit non-zero, and with implications for the long-term abundance of the species in Wales), may support decision making in this sector.

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Ethical Statement

The work presented here complies with the standards and procedures laid down by Welsh and UK legislation, and with the policies for ethical and reproducible research of the British Trust for Ornithology and Cardiff University. The authors have no conflicts of interest to declare.

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The 1923 Golden Eagle record from Meirionnydd

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In his review of birds in Meirionnydd, Pritchard (2012: p.76) briefly mentioned that a Golden Eagle *Aquila chrysaetos*, a rare non-breeding bird in Wales since before 1850, was recorded “near Llanuwchllyn in September 1923”. By implication, he suggested it might be a natural occurrence, as he then went on to state that “More recent records are likely to be escaped birds.”, an opinion that was confirmed subsequently in Pritchard (2021). What is not made clear is that this post-juvenile bird, seemingly an older sub-adult, is now a specimen held by the National Museum of Wales, Cardiff (NMW.Z.1945.165, see Fig. 1), which further holds a file of letters (ref. 45/165) concerning the circumstances of its occurrence and acquisition. This note provides a summary of their contents.

In March 1945, W. Miall Jones, Aberystwyth, wrote to the National Museum of Wales stating that “I have had a Golden? Eagle given me. It is mounted in a case. It was caught in a steel trap at Llanuwchllyn, Merioneth in Sep 1923 and was given me by Dr Davies [grandfather of TH], Bala, Merioneth” and offering to donate it to the museum. This offer was accepted by the museum by return of post. In June 1945, the museum wrote a further letter stating they were glad to acknowledge arrival of the mounted Eagle at the museum. This letter continued: “Regarding the official acknowledgement of the donation, I gather from your original letter and from one sent to Dr. Peate some time ago that this should go to Dr. Davies at Bala from whom you were kind enough to collect the bird on our behalf – perhaps you will confirm this.” In reply, Miall Jones confirmed that he presented the specimen with Davies’ permission and thought he would be glad of an official acknowledgement.

Dr Iorwerth C. Peate was the founder of St. Fagans National Museum of History, Cardiff, which opened in 1948, had previously studied and worked in Aberystwyth and Meirionnydd respectively, and had clearly also been involved in correspondence relating to the eagle specimen. Considerable additional detail concerning the taking of the specimen is available in a subsequent two-page hand-written document, annotated “From Dr A. L. Davies, Crud y Gwynt, Bala”, “Notes received from I.C. Peate, Nov. 1945” and “File under Miall Jones.” Much of this, which includes “a whole-hearted apology for the long delay in answering your letter”, is clearly from a letter written by Davies, but presumably with contributions from Peate interpolated. The relevant core of the document comprises the following:

“It [the eagle] was caught soon after the 1914-18 war on Buarthmeini mountain, Llanuwchllyn, by the tenant Watkin Jones. Buarthmeini is on the road from Llanuwchllyn to Trawsfynydd – two miles nearer Llanuwchllyn than Blaenlliw. Watkin Jones while “walking” his mountain near the house saw his dogs barking at an object on the ground. It was the eagle, unable to rise. He killed it with his stick. He then saw that one of the eagle’s middle “talons” was caught in an iron trap (you can see its mark). Watkin Jones found in a few days’ time that the trap was the property of General Vaughan of Nannau – his keeper had placed it to catch vermin on one of the Nannau hills, and the eagle had taken it fixed to its talons as far as Buarthmeini While I live, I shall believe that I saw a Golden Eagle flying above



Figure 1. The Golden Eagle specimen collected from Meirionnydd in September 1923 and now in the National Museum of Wales, Cardiff (NMW.Z.1945.165).

Llanuwchllyn. A few days before it was caught, I was fishing between two hillocks above Buarthmeini and the biggest bird I ever saw flew across from the one to the other!" The letter concludes by noting that Watkin Jones offered the specimen to Dr Davies and that it was sent to the Liverpool taxidermist William Cox to be mounted.

On the museum record for the specimen, it is noted that it had originally been housed in a case with a label written in Welsh; however the case and label are no longer present, and the specimen is now free-standing. The details of the label were "Daliwyd a'i droed mewn trap. Roedd cipar nannau wedi gosod trap at vermin. Ehedodd yr Eryr ar trap wrth ei droed i Waun Griolen, lle daliwyd ef gan Watkin Jones ac Evan J. Hughes, Buarth-y-Meini, Llanuwchllyn - Mis Medi, 1923." This translates as "It was caught by its foot in a trap. The Nannau Keeper had set a trap for vermin. The Eagle flew with the trap attached to its foot to Waun Griolen where it was caught by Watkin Jones and Evan John Hughes, Buarth-y-Meini, Llanuwchllyn – September, 1923."

The upland Nannau estate, with principal mansion nearly 250 metres above sea level, had until 1950 remained in the same family for c.850 years; when it was broken up and sold off in the 1960s and 1970s, the estate totalled about 1,400 hectares (Williams 2016, https://en.wikipedia.org/wiki/Nannau,_Wales). Much of the former estate appears to have comprised upland grazing land with some afforested areas, and it seems improbable that this has altered substantially over the past century. Exactly how far the Golden Eagle flew after getting caught in the trap is unclear, but possibly as much as a few kilometres.

On a more general note, the above account re-emphasises the potential of museum natural history collections and their associated archives to throw light on historical records of rare birds. It remains important that whenever a rarity is found dead, both it and associated information on its discovery are offered to an actively curated museum, notably a national one, that can preserve them in perpetuity and make them available for scientific study (Prys-Jones 1999).

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